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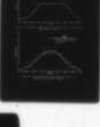
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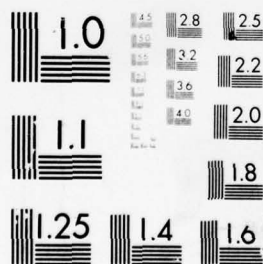
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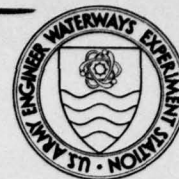
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WATER QUALITY EVALUATION OF PROPOSED TREXLER LAKE JORDAN CREEK, PENNSYLVANIA

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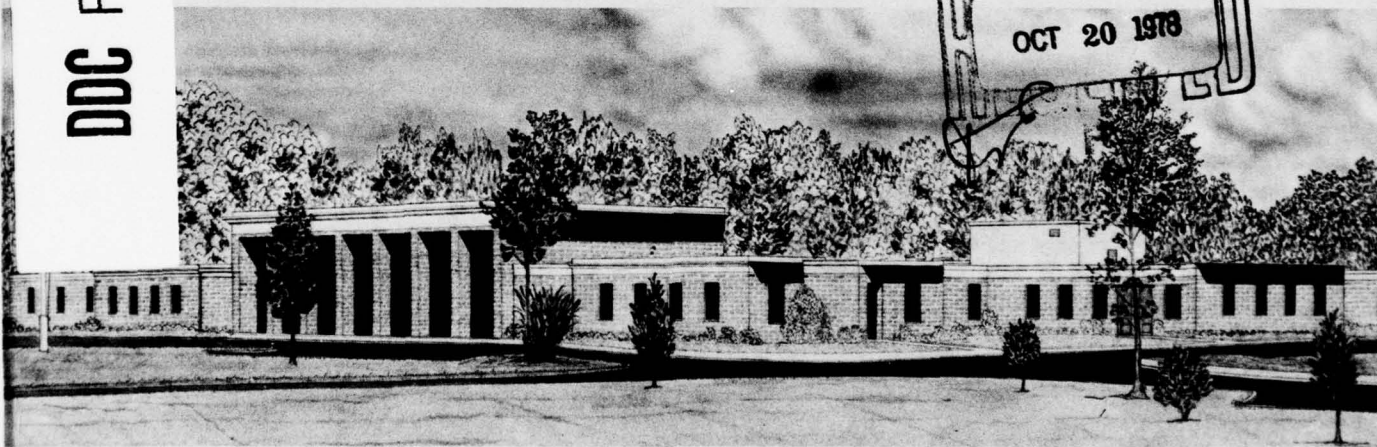
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August 1978

Final Report

Approved For Public Release; Distribution Unlimited



Prepared for U. S. Army Engineer District, Philadelphia
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20. ABSTRACT (Continue on reverse side if necessary and identify by block number) The water quality of proposed Trexler Lake, Pennsylvania, was evaluated with respect to its eutrophication potential and water quality criteria and standards appropriate for project purposes. Study approaches included review of existing water quality data taken on the four tributaries to the proposed project; review of existing data on surrounding impoundments; algal bioassay analyses on water samples taken in the proposed tributaries and in one surrounding lake; (Continued)		

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20. ABSTRACT (Continued).

CONT. mathematical simulations; and nutrient loading analyses. →

Water quality data from Mill, Lyon, Switzer, and Jordan creeks indicated no major water quality problems. Nitrate concentrations were high but did not exceed standards for water supply. Fecal coliform counts exceeded 200 colonies/100 ml during periods of high runoff. Local problems on Mill Creek can be expected under low flow because of effluents from the Heidelberg Heights sewage treatment plant.

The surrounding lakes can be classified as mesotrophic or mildly eutrophic, and none were able to meet the state standard of 5 mg/l for dissolved oxygen. Proposed Trexler Lake is expected to be similar in trophic status and in dissolved oxygen content.

→ Algal bioassays were conducted on water samples taken in Mill, Lyon, Switzer, and Jordan creeks and in Beltzville Lake. All samples were phosphorus limited, and all of the orthophosphate was available for algal growth.

Mathematical simulations included both watershed and reservoir models. The Hydrocomp Simulation Program was used to predict inflows from each of the four tributaries. A reservoir ecological model was used to predict the trophic status of the impoundment and the effects of various reservoir regulation schemes on the water quality. Because insufficient water quality data existed on the project to quantify loadings accurately and on surrounding impoundments to calibrate the model, Monte Carlo simulations were used to specify water quality update data and coefficients.

The mathematical simulations indicated that the downstream temperature objective could be met provided the selective withdrawal structure is redesigned to have six ports evenly distributed in two wet wells. Simulated algal concentrations were 5 to 11 mg/m³ chlorophyll *a* or similar to the concentrations found in surrounding lakes. The hypolimnion was predicted to be anoxic for about 1 month. No problems with fecal coliforms exceeding standards were predicted, but intermittent problems in the headwater regions are probable.

Nutrient loading analyses were performed using the concepts of Vollenweider, Dillon and Rigler, and Larsen and Mercier. All three methods predicted proposed Trexler Lake to be eutrophic. Chlorophyll *a* concentrations estimated from the average in-lake phosphorus concentrations were equal to or slightly higher than the concentrations found in surrounding lakes.

→ The proposed Trexler Lake is expected to be in the mesotrophic or early eutrophic stage and exhibit strong thermal stratification during the summer months with a 1-month period of hypolimnetic anoxia during the fall. The project purposes are not expected to be vitiated by water quality although the lake is expected to be anoxic for a short time during late summer.

Appendix A describes the algal assay procedures. Appendix B discusses the Hydrocomp simulation of Jordan Creek drainage basin. Appendix C presents the initial conditions, coefficients, and updates for mathematical ecological simulations. Appendix D lists the coefficient references.

PREFACE

This study was conducted by the Environmental Laboratory (EL) of the U. S. Army Engineer Waterways Experiment Station (WES), Vicksburg, Mississippi, for the U. S. Army Engineer District, Philadelphia (NAP). The project was authorized by Intra-Army Order for Reimbursable Services No. NAPEN 77-3 dated 28 October 1976. Model refinements, modifications, and Monte Carlo simulations were accomplished as part of the Environmental Impact Research Program and the Environmental and Water Quality Operational Studies Research Program of the Office, Chief of Engineers, U. S. Army. Dr. John Burnes, NAP, monitored the project.

This report is an evaluation of the water quality expected in the proposed Trexler Lake relative to its eutrophication potential and to water quality criteria and standards appropriate for the project purposes.

The research was conducted under the direct supervision of Mr. D. L. Robey, Chief, Ecosystem Modeling Branch (EMB), and under the general supervision of Drs. R. L. Eley, Chief, Ecosystem Research and Simulation Division, EL, and John Harrison, Chief, EL. Drs. D. E. Ford and K. W. Thornton, EMB, served as principal investigators. Dr. A. S. Lessem and Ms. Connie Stirgus, EMB, participated in the study and, along with Ms. Carol Henry, EMB, assisted in the data analysis and model simulations. Dr. Eugene R. Perrier and Ms. Jane Harris, EMB, modeled the Jordan Creek drainage basin using the Hydrocomp Simulation Program and prepared Appendix B. Drs. J. Barko, Ecosystem Processes Research Branch, and R. H. Kennedy, EMB, reviewed the draft report.

The draft report was also reviewed by representatives from the U. S. Army Engineer Division, North Atlantic, NAP, and Delaware River Basin Commission, who met with representatives from WES at NAP on 10-11 January 1978.

Mr. J. L. Barker, U. S. Geological Survey, Harrisburg, Pennsylvania; Mr. R. Boardman, Pennsylvania Department of Environmental Resources, Harrisburg; and Mr. J. Angello, New Jersey Department of Environmental Protection, Division of Fish, Game, and Shell Fisheries,

Lebanon, provided assistance in data compilation.

Ms. Leslie A. Gardner, Utah Water Research Laboratory, Utah State University, Logan, conducted the algal bioassay analyses and prepared Appendix A.

Director of WES during the conduct of this study and the preparation and publication of this report was COL J. L. Cannon, CE. Technical Director was Mr. F. R. Brown.

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WATER QUALITY EVALUATION OF PROPOSED TREXLER LAKE
JORDAN CREEK, PENNSYLVANIA

PART I: INTRODUCTION

1. The proposed Trexler Lake project as authorized would consist of an earth- and rock-fill dam and a reservoir that would provide water supply, recreation, and flood control benefits, which would contribute 62, 21, and 17 percent of the project benefits, respectively. The proposed project would be located on Jordan Creek in Lehigh County, about 12 km northwest of the city of Allentown in southeastern Pennsylvania (Figure 1). In addition to Jordan Creek, major tributaries to the proposed reservoir include Mill, Switzer, and Lyon creeks. The conservation pool would be at elevation 150.3 m* msl and would impound approximately $5.06 \times 10^7 \text{ m}^3$ of water with a surface area of 493.0 ha. This project would create a lake 31.4 m deep at the dam and extending 13.9 km upstream. At the full pool elevation of 153.6 m msl, $6.86 \times 10^7 \text{ m}^3$ of water would be impounded, inundating approximately 600 ha.

2. The objective of this study was to evaluate the water quality and eutrophication potential of proposed Trexler Lake relative to its project purposes. A combination of techniques including comparisons with surrounding impoundments, algal bioassays, mathematical simulations, and nutrient loading analyses was used. While the individual techniques have inherent assumptions and limitations, their combined use provides corroborative and complementary information. This approach has been used in other water quality studies (Thornton et al. 1976; Hall et al. 1977) and is described in detail by Thornton et al. (1977b).

3. This report presents results of a water quality evaluation of proposed Trexler Lake. The assumptions, limitations, and results of each technique are presented separately. Predictions from the

* Elevations cited herein are in metres referred to mean sea level (msl); sampling elevations are referred to base of reservoir.

different techniques are evaluated and compared based on the appropriate assumptions and limitations to determine the water quality and trophic status of proposed Trexler Lake. The specifics of the algal bioassay analyses and the Hydrocomp simulations are described in Appendixes A and B, respectively. Model coefficients, updates, and initial conditions are summarized in Appendix C. Coefficient references are listed in Appendix D.

PART II: EXISTING WATER QUALITY DATA

4. One of the best approaches for predicting the water quality in a proposed project is to analyze existing riverine and impoundment water quality data and to extrapolate the results to the proposed impoundment.

Stream Data

5. Table 1 summarizes all known water quality data taken at the proposed Trexler project site. The specific field surveys included Allentown-Bethlehem Bi-City Health Bureau (Everett 1976), U. S. Geological Survey (USGS) (1972-1975), U. S. Environmental Protection Agency (Kaeufer 1973), Pennsylvania Department of Environmental Resources (unpublished data for 1966, 1971, and 1972), and Pennsylvania Department of Health (unpublished data for 1966). Only data taken in Jordan Creek and tributaries at or above the USGS gage at Schnecksville (the approximate damsite) were considered (Figure 1). Data from the Heidelberg Heights sewage treatment plant effluent on Mill Creek were specifically excluded, but data at the downstream gage on Mill Creek, after the effluent mixed with the stream waters, were included.

6. Several of the water quality constituents summarized in Table 1 exhibit large variations or ranges (e.g., total organic carbon (TOC), total coliforms, $\text{NO}_3\text{-N}$, pH). A few high or low values (i.e. outliers) usually account for these extended ranges. Differences between means and medians for any one constituent are also attributable to outliers.

7. A more representative picture of the expected water quality in the proposed Trexler Lake may be found by considering only the data taken at the damsite (Table 2). These data represent a volume-weighted composite of all the water entering the impoundment. These data are also presented in the form of scatter plots in Figures 2-15 for all parameters except organic nitrogen (N), dissolved oxygen (DO), pH, and biochemical oxygen demand (BOD5). There is little or no relationship

between most constituents and flow. Some outlying data points are visible (Figures 3, 5, 8, 11, 14, and 15). There is, however, less scatter in these data than in the data summarized in Table 1.

8. Care must be taken in extrapolating these data from the stream to the impoundment because of the change in flow regime. As the water flows from a riverine environment into a lake environment, its depth increases and travel times and turbulence decrease. There is less surface aeration, more decay per unit distance traveled, and more opportunity for particulate matter to settle out. Therefore, water quality constituents that decay with time and/or adhere to particulate matter should be found in smaller concentrations in the impoundment than in the stream, provided no other sources exist.

9. The existing water quality data from the tributaries to proposed Trexler Lake indicated no major water quality problems. Phosphorus (P) and ammonia concentrations were low while nitrate was high (Table 2). However, the nitrate concentrations did not exceed U. S. Environmental Protection Agency (EPA) recommended water quality criteria (EPA 1976) for water supply (i.e. 10 mg/l as N). Since nitrate concentrations in the impoundment should be lower than in the tributaries, no problems are expected in the impoundment. The median N/P ratio of 100 (i.e., total N/total P = 3.05/0.03 from Table 2) indicated potential phosphorus limitation of phytoplankton growth. Since $\text{PO}_4\text{-P}$ concentrations were low (Table 2), proposed Trexler Lake would probably not be able to support obnoxious algal blooms.

10. Local problems with fecal coliforms may also occur whenever the input is larger than the state standard of 200 colonies per 100 ml. The median value of 280 and maximum of 9600 fecal coliforms per 100 ml (Table 2) in the inflow indicate that this standard could be exceeded. However, impoundment of the streams will result in longer residence times and more opportunity for settling and decay.

11. The range of pH values in Table 2 is large. There does, however, appear to be a relationship between pH and the time of year. During the winter and periods of high runoff, pH values tend to be less than 7. This is expected because of the acidic nature of the

soils in the watershed above the damsite (Appendix B). The high pH's occur during periods of low flow and high biological activity. Typical chlorophyll a values during these periods are 50 µg/l, indicating the potential for primary production to increase and deplete the free carbon dioxide pool, thereby elevating the pH.

12. In order to evaluate the impact of each of the tributaries, mean values for several water quality constituents are compared in Table 3. As expected, the impact of the Heidelberg Heights sewage treatment plant on Mill Creek was reflected in the higher mean values for TOC, NO₃-N, total N, total P, and PO₄-P. Some local problems with algal blooms may occur on this branch of the proposed impoundment under low-flow conditions from the phosphorus inputs of the Heidelberg Heights sewage treatment plant. Lyon Creek was characterized by higher counts of fecal coliforms and fecal and total streptococcus bacteria. These can be attributed to a barnyard upstream from the sampling station. Local problems may also occur in this branch.

Impoundment Data

13. Three existing U. S. Army Corps of Engineer (CE) impoundments in the general vicinity of proposed Trexler Lake are Beltzville, Prompton, and Francis E. Walter lakes. Their major morphometric characteristics are compared with proposed Trexler Lake in Table 4. Since Beltzville is located approximately 22 km north of the proposed project and since it is also very similar in morphometry, it was a likely candidate for comparison. It should be noted, however, that Beltzville is long and narrow with one major arm and tributary while Trexler is dendritic with four major arms and tributaries. The two other impoundments are located further away and are not morphometrically similar. In addition, Prompton Lake has been subjected to copper sulfate treatment and hypolimnetic aeration, making it undesirable for comparison. Other impoundments considered in the study are listed in Table 5.

14. Generally, all of the surrounding impoundments were of good water quality. Isolated problems with turbidity and algal blooms did,

however, occur in the headwater regions following rainstorms and other periods of high runoff. The runoff was also slightly acidic in the three CE impoundments. All three CE impoundments met state standards for body contact recreation.

15. All of the surrounding impoundments experienced low DO in the hypolimnion. None were able to meet the state standard of 5 mg/l for all points, which includes the hypolimnion of lakes. Typical DO profiles for Beltzville Lake are shown in Figures 16 and 17. The hypolimnion usually went anoxic by October. In the shallower lakes, anoxia occurred as early as July.

16. The major species of phytoplankton found in the surrounding lakes are shown in Table 6. There was, however, no definite pattern of algal succession. Diatom blooms were just as common in July as in May and blue-green algae were just as likely to occur in October or November as August. The change in species composition for Conewago Lake is shown in Figure 18. *Ceratium* was found in sufficient numbers in some of the lakes to cause taste and odor problems with water supply.

17. Typical chlorophyll a concentrations found in the surrounding impoundments are summarized in Table 7. If the recommendation by the National Eutrophication Survey (Gakstatter et al. 1975) that a concentration of 10 mg/m³ chlorophyll a separates mesotrophic from eutrophic lakes is assumed to be valid, then the surrounding impoundments would have to be classified as mesotrophic or in the early eutrophic stage. This conclusion is consistent with the findings of the National Eutrophication Survey (EPA 1975) for Beltzville Lake and of Ott et al. (1973) for Conewago Lake.

18. Proposed Trexler Lake is expected to have general water quality characteristics similar to these surrounding impoundments, including temperature stratification, anoxic hypolimnion, and trophic status. Of the surrounding impoundments considered, Beltzville Lake is, perhaps, the best analog to proposed Trexler Lake because of similarities in morphology, geographical location, and general watershed characteristics.

PART III: ALGAL BIOASSAYS

19. Algal bioassays were conducted on water samples collected at the proposed Trexler project site to determine the nutrient(s) potentially limiting phytoplankton growth and the availability of the nutrients for plankton uptake. The analyses were performed by the Utah Water Research Laboratory, Utah State University, Logan (Appendix A). This information was required to modify update data and select appropriate coefficients for the mathematical ecological simulations.

20. An assumption of the mathematical ecological model used in this study was that all nutrient inputs were in a 100 percent available form for phytoplankton uptake. If only a fraction of the nutrient, say phosphorus, were available for growth, then the input data should be modified to reflect this availability. In addition, model coefficients, such as half-saturation coefficients, must be selected to reflect nutrient concentrations and the limiting nutrient. The various nutrient-loading models used to predict trophic status also require a priori knowledge of the limiting nutrient.

21. Water samples were collected by U. S. Army Engineer Waterways Experiment Station (WES) personnel at the proposed project site on 30 March and 21 June 1977 and at Beltzville Reservoir on 30 March. The specific sampling locations were

- a. Near the bank at the USGS gaging station on Jordan Creek near Schnecksville (sample GS800B).
- b. Center of stream at the USGS gaging station on Jordan Creek near Schnecksville (sample GS800C).
- c. USGS sampling station on Mill Creek near Schnecksville (sample GS770).
- d. USGS sampling station on Lyon Creek at Lyon Valley (sample GS738).
- e. USGS sampling station on Switzer Creek near Pleasant Corners (sample GS700).
- f. USGS sampling station on Jordan Creek near Pleasant Corners (sample GS695).
- g. Beltzville Lake, 2.5 km upstream from the dam (sample Belt 1).

- h. Beltzville Lake, 4.0 km upstream from the dam (sample Belt 2).
- i. Pohopoco Creek above Beltzville Lake (sample Belt 3).*

22. Water for the creek samples was taken from different depths and across the channel to obtain composite samples. On Beltzville Lake, composite samples were pumped from the top 8 m. The lake temperature ranged from 5°C at 9 m to 8.5°C at the water surface.

23. Water samples were taken twice during 1977 to determine seasonal variations. The 30 March date was selected to represent spring runoff while 21 June was selected to represent summer base flow. Preliminary flow data from USGS indicated that March 1977 was the wettest March on record at the Schnecksville gage and that June 1977 was drier than average. Flows at the Schnecksville gage on the two dates were 2.63 and 0.74 m³/sec, respectively. In 1977, most of the spring runoff occurred before 30 March. The spring sample, therefore, characterized water on the trailing side of the spring runoff hydrograph. The quality of this water may not be the same as that found at the peak or on the rising side of the hydrograph because many water quality parameters do not load proportionally with flow. Rain showers over the watershed on the afternoon of 20 June resulted in the flow at Schnecksville increasing from 0.37 m³/sec on 19 and 20 June to 0.74 m³/sec on 21 June, when the second set of samples were taken. These later samples were probably characteristic of summer runoff because the creek waters were turbid at the time of sampling.

24. All samples were preserved with ice and shipped by airfreight to Utah Water Research Laboratory. All samples arrived intact except Belt 3, which was damaged. Since this was the only Beltzville inflow sample, no conclusions can be drawn between conditions in the inflow and in the lake. Upon arrival at the Utah Water Research Laboratory, the water samples were filtered and chemically analyzed. The algal bioassays were performed according to EPA (1971) procedures. The

* Sample was damaged in transit.

results are briefly described in the following paragraphs and presented in detail in Appendix A.

25. All nutrient concentrations except $\text{NH}_3\text{-N}$ were found to be higher in the second set of samples than in the first. This finding was not unexpected because of the rain on the day preceding the June sampling. The N/P ratios of all the stream samples except Mill Creek were 190 or greater, indicating phosphorus limitation. The ratios for the two Mill Creek samples were 22 and 20 for 30 March and 21 June, respectively. Since these values were close to the optimum level ($\text{N/P} \approx 15$) required for growth, spiking tests were necessary to determine the limiting nutrient. In the two Beltzville samples, the N/P ratios were 146 and 160, also indicating phosphorus limitation. These two samples were also characterized by the lowest soluble inorganic nitrogen concentrations.

26. Briefly, spiking tests consist of injecting a nutrient into a sample and measuring the response of the algae. If the algae respond with growth to the injection or spike, then that nutrient is probably limiting. Spiking tests were performed with $\text{NH}_3\text{-N}$, $\text{PO}_4\text{-P}$, NAAM (nutrient algal assay medium),* and trace elements. $\text{NH}_3\text{-N}$ was substituted for $\text{NO}_3\text{-N}$ because of high $\text{NO}_3\text{-N}$ concentrations in the samples.

27. The spiking tests showed that phosphorus was limiting in all of the samples taken on 30 March. The Mill Creek sample was able to support a significant biomass increase without treatment. With spiking, phosphorus became limiting. In the Beltzville samples, the algae responded to phosphorus, but nitrogen eventually became limiting. All of the other samples showed that phosphorus was the nutrient limiting growth.

28. The results from the second set of samples (21 June) were slightly different. The nutrient concentrations were so high in the Mill Creek sample that spiking resulted in a decrease in growth. It was concluded that there was no limiting factor. In the other samples, phosphorus was limiting, but the response to $\text{PO}_4\text{-P}$ was less than the

* NAAM is described in Appendix A, Table A2.

response to NAAM. This result indicated that one of the other nutrients eventually became limiting. The algal bioassays also indicated that all of the $\text{PO}_4\text{-P}$ was in a form available for growth.

29. In summary, the algal bioassays indicated that phosphorus would be potentially limiting in proposed Trexler Lake. In addition, the phosphorus concentrations were sufficiently low to prevent much algal growth except in Mill Creek. These general findings are in agreement with the bioassays performed by the National Eutrophication Survey on Beltzville Lake in 1973 (EPA 1975).

PART IV: MATHEMATICAL SIMULATIONS

30. A research version of the Water Quality for River-Reservoir Systems (WQRRS) reservoir model was used in this study. The model was originally modified for the Hydrologic Engineering Center, Davis, California, by Water Resources Engineers (WRE), Walnut Creek, California. This model has been applied to a variety of CE project studies in the past (Thornton et al. 1976; Thornton et al. 1977a; Hall et al. 1977; Ford et al. 1977; and others). Since the 1977 documentation for the WQRRS model (Hydrologic Engineering Center 1977), several significant modifications have been incorporated into the model by the Environmental Laboratory at WES. A partial listing of these is presented in Table 8. These modifications were made to produce more realistic simulations of reservoir ecosystems. Before results of the model simulations made for Trexler Lake are discussed, it is important to examine and understand the major assumptions and limitations of the WQRRS model since these play an important role in the interpretation process.

Major Assumptions and Limitations of the WQRRS Model

31. Major assumptions and limitations of the WQRRS model are as follows:

- a. A reservoir can be represented by a vertical series of one-dimensional horizontal slices. This implies that only the vertical dimension is retained during computation. Each horizontal layer is assumed to be completely homogeneous. Therefore, all isotherms and all isopleths of water quality constituents such as DO, nutrients, and algae or other biota are parallel to the water surface both laterally and longitudinally. In addition, all inflow and outflow quantities and concentrations are instantaneously dispersed and homogeneously mixed throughout each horizontal layer from the headwaters of the impoundment to the dam. It is not possible, therefore, to look at longitudinal variations in constituents such as coliforms or phytoplankton.
- b. In general, the vertical dimensions are specified for the deepest portion of the reservoir, and the model results will be most representative of conditions in this area.

Since this area is usually near the structure, it is difficult to draw conclusions on water quality conditions expected to occur in coves, embayments, or headwater area. Similarly, the specification of initial conditions is for the deepest area of the reservoir.

- c. Vertical placement of inflowing water within the impoundment is determined by temperature only. The density of an inflow is determined from its temperature, and it is placed into the horizontal zone that has the same density. Contributions to the density of the inflow by suspended and dissolved solids are not currently included in the model. These contributions may not be significant in this study. In the headwaters, both deposition and inflow mixing occur. The effect of each of these processes is to decrease the contribution of suspended solids to the density of the inflow. In addition, calculations show that the density contribution of suspended solids concentrations up to 1000 mg/l is less than the density change resulting from a 1°C drop in temperature at 25°C.
- d. Internal dispersion of thermal energy and mass is accomplished by an effective diffusion mechanism that combines the effects of molecular diffusion, turbulent stirring and mixing, and thermal convection. The transport is therefore assumed to be proportional to an effective diffusion coefficient and a concentration gradient. It is important to note that although the diffusion gradient among layers is based on the concentration differences of the individual constituents such as DO or nitrate, the effective diffusion coefficient is always based on temperature. In many instances, mass diffusion coefficients may not be equivalent to thermal diffusion coefficients.
- e. The dynamics of each chemical and biological component can be described by conservation of mass and the kinetic principle. The mass of elements such as carbon, oxygen, nitrogen, and phosphorus is accounted for by considering the inflows, outflows, and internal changes in the form of the elements. The kinetic principle implies that these internal changes occur through rate processes such as uptake, decay, respiration, etc.
- f. The chemical and biological rate processes occur in an aerobic environment. While the WQRRS model does have some simple default algorithms that occur when DO is zero, the model predictions are not realistic under anoxic or zero DO conditions. The model algorithms were developed to simulate the rate processes and reactions occurring under aerobic conditions. This limitation results in an inability to simulate the buildup of a DO deficit under

anaerobic conditions. Consequently, the model may predict the timing and rate of DO increases following the end of anaerobic conditions to be greater than might actually occur. Also, changes in the solubility and formation of various chemical species and interactions between sediment and water under anaerobic conditions cannot be simulated.

- g. The model does not contain an ice cover algorithm. Since conditions under an ice cover cannot be simulated, model predictions are limited to ice-free periods.

Selection of Study Years

32. The U. S. Army Engineer District, Philadelphia (NAP), determined that 3 years representing dry, wet, and average conditions should be simulated. These were selected to be 1969, 1973, and 1974, respectively, based on 30 years of flow record at the USGS gage at Allentown, Pennsylvania. Since flow records were begun at the Schnecksville gage in 1966, only years after 1966 were considered possible candidates. Emphasis was also placed on years following 1971, since they were characterized by the most water quality data. Based on the logarithmic frequency analysis of the 30 years of flow record, 1969 fell within the lowest 15 percent and 1973 within the highest 15 percent flow category, while 1974 was near the mean flow. It was necessary to include 1969, for which there was minimal water quality data, since there were no dry years after 1971.

33. In Table 9, the mean monthly flows for the three study years are compared with the 30-year monthly averages based on logarithmic analysis. The year 1969 was characterized by below-average flows from January through June and November through December. July and August were above average, making the seasonal distribution approximately constant. In 1973, the winter and spring were wet and the summer was about average. The year 1974 followed the general trends exhibited by the long-term average: high flows in the winter and spring and low flows during the summer.

Data Requirements

- 34. Data requirements for the ecological model can be categorized

into initial conditions, coefficients, and updates.

Initial conditions

35. Simulations were started on 1 April (Julian day 91) with isotropic conditions. The specific values are summarized in Appendix C. With the exception of temperature and DO, all other initial conditions were assumed invariant for the three study years. Temperatures were taken to be equal to the average equilibrium temperature for the preceding 10-day period (i.e. days 82-91). For the study years 1969, 1973, and 1974, the initial temperatures were 6.0, 9.0, and 5.0°C, respectively. The DO concentrations were assumed to be saturated at these temperatures.

36. A comparison of inflow and in situ concentrations of the major nutrients, total dissolved solids (TDS), and pH showed that they were similar in Prompton, Beltsville, and Francis E. Walter lakes during April. It was therefore assumed that the initial conditions for these constituents in the proposed Trexler Lake would be similar to those measured in Jordan Creek at the beginning of the simulation. For such constituents as fecal coliforms, dissolved organics, and detritus that are expected to decay or settle out with time and with a change in the flow regime from river to reservoir, initial conditions were selected to be less than those measured and to correspond to values found in surrounding lakes. Initial standing crop estimates for the biological organisms were also selected to be of similar magnitude to those found in surrounding lakes.

Coefficients

37. Coefficients required to simulate Trexler Lake ranged from those that are physically definable and well documented in the literature to those that are difficult to measure in either the laboratory or the field and must be quantified, based primarily on the experience of the investigator and calibration runs. The coefficients used in the base simulations are listed in Appendix C.

38. The coefficients that determine the hydrothermal regime were determined by calibrating the model on Beltsville Lake. Since Beltsville is located only 22 km from Trexler and since the two

impoundments have similar morphometry (Table 4), the mixing and heat budget coefficients should be similar. The calibration is discussed in detail in a later section on thermal simulations.

39. Since the ecological model has the capability to simulate only two types of algae, the species of phytoplankton listed in Table 6 had to be aggregated into two groups. Lewis (1977) found strong correlations in net growth rate between abundant species at intradivisional levels and between morphologically similar species. Using both criteria, the blue-greens and greens were combined in the ALGAE 1 compartment, and the diatoms and desmids were put into the ALGAE 2 compartment. The desmids, *Pediastrum* sp. and *Stauroastrum* sp., were included in the diatom compartment based on morphological and functional criteria while the other greens, typical of eutrophic impoundments, were included with the blue-green algae for similar reasons. *Ceratium* was not included in the aggregation because it exhibits heterotrophic properties and cannot be simulated by the model in its current form. It is, however, probable that *Ceratium* will be found in Trexler Lake in numbers similar to those found in surrounding impoundments and may at times produce taste and odor problems in the water supply.

40. Growth, respiration, and settling rates and nitrogen, phosphorus, carbon, and light half-saturation coefficients for the compartmental assemblages based on the species listed in Table 6 were compiled from the literature (Appendix D). These were weighted by dominance and the percent time the species were found in the surrounding impoundments to obtain composite growth rates for each aggregation.

41. The model considers only one type of zooplankton and benthos. The following zooplankton were found in the surrounding impoundments: Rotifera: *Keratella* sp., *Polyarthra* sp., *Kellicottia* sp., and *Trichocerca* sp.; Cladocera: *Daphnia* sp.; and Copepoda: *Cyclops* sp. Since there was little information on benthos in the area, it was assumed to be similar to that found in lakes of similar latitude. It was, therefore, assumed that chironomids and tubificids dominated the profundal region. Coefficients were selected to represent these compositions.

42. The fishery compartment in the model is capable of simulating planktivores, benthos feeders, and piscivores. Based on information in the Trexler Environmental Impact Statement (NAP 1974a) and Leidy and Jenkins (1977), the planktivores were assumed to be minnows and suckers; the benthos feeders were assumed to be carp, suckers, and sunfish; and the predators were assumed to be black and temperate basses, sunfish, and pickerel. Coefficients for these compositions were taken from Leidy and Jenkins (1977).

43. Other coefficients such as decay rates for BOD5 and $\text{NH}_4\text{-N}$, etc., were taken from Sawyer and McCarty (1967), Kittrell and Furfari (1963), previous simulations, and other literature cited in Appendix D.

Updates

44. Updates refer to model inputs that vary with time. These include meteorological data, flow data, and inflowing constituent concentrations. To minimize the quantity of data required, the four major tributaries were combined. This simplification makes it possible to use the larger data base that exists at the USGS Schnecksville gage. No precision is sacrificed by this simplification because the model is one-dimensional, and all inflowing constituents are instantaneously dispersed throughout each horizontal layer (paragraph 31).

45. The meteorological data were obtained from the Class A weather station at Allentown, Pennsylvania, located approximately 12 km southeast of the proposed project site. The 3-hr data were averaged over 24 hr for daily updates.

46. Daily inflows to the reservoir were obtained from the gage at Schnecksville. The 1969, 1973, and 1974 daily discharges are shown in Figures 19-21, respectively. Flows were also generated for each of the four tributaries using the Hydrocomp Simulation Program (Appendix B). The inflows were routed through the reservoir by NAP for four regulation schemes:

- a. 39-cfs ($1.1\text{-m}^3/\text{sec}$) diversion, 6-cfs ($0.17\text{-m}^3/\text{sec}$) minimum release ($1.27\text{ m}^3/\text{sec}$).
- b. 54-cfs ($1.53\text{-m}^3/\text{sec}$) diversion, 6-cfs minimum release ($1.70\text{ m}^3/\text{sec}$).
- c. 58-cfs ($1.64\text{-m}^3/\text{sec}$) Trenton supplement.

d. 110-cfs ($3.11\text{-m}^3/\text{sec}$) Trenton supplement.

The diversion in the first two regulation schemes corresponds to the water supply yield. The last two schemes correspond to the situation in which the water supply benefits are not immediately utilized and the releases are used to supplement the flow in the Delaware River at Trenton. The specific operational schedule for the selective withdrawal structure is discussed in a later section on thermal simulations (paragraph 54).

47. Daily water temperatures at the Schnecksville gage were generated from daily air temperatures using residuals between measured temperatures and generated first-order harmonics. That is, a seasonal cosine cycle or first-order harmonic was fitted to the daily air temperature measured at Allentown by the method of least squares. The first-order harmonics for the years 1972-1975 are shown in the following tabulation:

Year	Equation	Standard Error
1972	$T_{ah} = 10.22 - 12.55 \cos [0.0172(\text{JD} - 21.2)]$	± 3.80
1973	$T_{ah} = 11.59 - 12.27 \cos [0.0172(\text{JD} - 22.8)]$	± 3.99
1974	$T_{ah} = 10.87 - 11.55 \cos [0.0172(\text{JD} - 17.4)]$	± 3.91
1975	$T_{ah} = 11.24 - 12.19 \cos [0.0172(\text{JD} - 24.0)]$	± 4.14

where

T_{ah} = harmonic air temperature, $^{\circ}\text{C}$

JD = Julian day

48. The air temperature residuals or differences between the measured temperature T_a and the generated seasonal curve were then formed. A seasonal cosine curve was similarly computed for all the water temperature data at the Schnecksville gage:

$$T_{wh} = 11.58 - 9.11 \cos [0.0172(\text{JD} - 22.3)] \quad \pm 2.58^{\circ}\text{C}$$

where T_{wh} = harmonic water temperature, $^{\circ}\text{C}$.

49. Water temperature residuals were then formed. Multiple regression was run on the water temperature residual for day I with the air temperature residuals for days I, I-1, I-2, and I-3. Retaining only terms significant to the 0.10 level, the water temperature for day I, $T_w(I)$, is

$$T_w(I) = T_{wh} + 0.1381 + 0.409 * T_{ar}(I-1) \quad \pm 2.05^\circ\text{C}$$

where $T_{ar}(I-1)$ = air temperature residual for day I-1 (i.e., $T_{ar}(I-1) = T_{ah} - T_a(I-1)$). The daily temperatures generated by the air temperature residuals have a slightly better standard error than temperatures obtained from the first-order harmonic (2.05 versus 2.58) and were therefore used in this study.

50. The DO in the inflows was assumed to be 100 percent saturated. The range of percent saturation for the USGS data for 1972-1975 was 93-100 percent. Diurnal DO measurements by EPA ranged from 81 to 136 percent saturation. The range of DO saturation in Mill Creek, the only tributary that did not show supersaturation, was 81-97 percent.

51. Since alkalinity data were limited, they were generated from the regression equation with flow:

$$ALK = 38.9(\text{Flow})^{-0.269}$$

where

ALK = alkalinity, mg/l

Flow = streamflow, m^3/sec

The coefficient of determination was 0.57. The regression equation is compared with data in Figure 22.

52. Since the reservoir will be a completely different system from the stream, it was assumed that there would be no inputs of algae and zooplankton from the tributaries to the reservoir that would remain viable. The stream inputs for these variables were considered to be included in the detritus updates. All other water quality updates except for BOD5, $\text{NO}_2\text{-N}$, and the 1969 updates were obtained from linear

interpolation between data. Updates for 1969, BOD5, and $\text{NO}_2\text{-N}$ were assumed constant at the median concentrations (Table 2) because little or no water quality data existed.

53. Several of the updates were varied in the Monte Carlo simulations (paragraph 96) to determine their sensitivity and impact on simulation results.

Thermal Simulations

54. Thermal simulations were used to calibrate the mixing and heat transfer coefficients required for the ecological model simulations and to evaluate the selective withdrawal capabilities of the project with respect to a downstream temperature objective.

Calibration of thermal structure

55. Beltzville Lake was selected as the prototype to calibrate the thermal structure of proposed Trexler Lake for the following reasons:

- a. The morphometric characteristics, including theoretical hydraulic residence time, are similar (Table 4).
- b. Since the two projects are located only 22 km apart, they should be exposed to similar meteorological conditions.

56. The thermal structure of Beltzville Lake in 1972 was simulated using data from Marcinski (1975). The simulation started on day 138 with a measured temperature profile. The calibration results are shown in Figure 23. Simulated surface temperatures were within 1°C of measured temperatures, and the location of the thermocline corresponded to measured field data. The fall overturn was also simulated to occur at the correct time. The high hypolimnion temperatures after day 201 resulted from excessive mixing accompanying the release of floodwaters following Hurricane Agnes (days 178-187).

Temperature objective

57. According to Commonwealth of Pennsylvania regulations (1971), the waters released from proposed Trexler Lake into Jordan Creek are not to increase water temperature by more than 2.8°C above natural

temperatures nor above 14.5°C overall. A downstream temperature objective that followed the natural temperature cycle of Jordan Creek up to 14.5°C was established. For natural temperatures above 14.5°C, the objective was set at 14.5 (Figure 24).

58. The natural temperature cycle was determined by fitting a first-order harmonic to all the temperature data for the Schnecksville gage by the method of least squares. The result was as follows:

$$T = 11.58 - 9.11 \cos [0.0172(JD - 22.3)]$$

where T = temperature, °C. This is the same equation used to generate inflow temperatures. Generated data are compared with actual data in Figure 25.

Selective withdrawal capabilities

59. The selective withdrawal capabilities of the Trexler project were evaluated using the thermal compartment of the ecological model and selective withdrawal routines developed by the WES Hydraulics Laboratory. Simulations were made to determine the feasibility of meeting the downstream temperature objective. NAP requested that the criteria for downstream releases also be applied to water supply releases.

60. The selective withdrawal structure described in the Trexler Lake Design Memorandum No. 7 (NAP 1974b) used separate and symmetrical wet wells for water supply and downstream releases. The specifics of the withdrawal ports are given in the following tabulation:

Port No.	Center-Line Elevation m	Size, m	Maximum Flow Capacity m ³ /sec
1	146.4	1.37 × 2.59	14.2
2	137.3	1.37 × 2.59	14.2
3	125.1	1.37 × 2.59	14.2
2 floodgates	121.8	1.37 × 1.98	80.1 (both)

61. The first simulation assumed that only one port per wet well would be operated to prevent blockage of flow from the upper port during

stratified conditions. The simulated outflow temperatures are compared with the temperature objective in Figure 26. The temperature objective was not met because flow from two ports was not allowed to blend. State standards were also violated in midsummer when the flow was switched from the top port to the second port. Release temperatures dropped by more than 10°C. Because of the depth of proposed Trexler Lake, additional ports would be required to operate the structure with only one open port per wet well and still meet state standards.

62. Better results were obtained when the flow from two ports was blended (Figure 27). The temperature objective was met except during the initial warming phase during the early spring period. The lower temperatures in the spring resulted from the top port being too far below the water surface.

63. Operating the structure with two open ports in the same wet well during stratification might result in difficulties in controlling the flow through each port. The heavier water flowing through the lower port would tend to block the flow of the lighter water through the top port, resulting in a highly unsteady pulsating flow. This design is also undesirable because of the large distance between ports, making it very difficult to use the structure for selective management of other water quality variables.

64. An alternative design is compared with the original design in Figure 28. This design consists of six selective withdrawal ports staggered in two wet wells. The specifics are given in the following tabulation:

Port No.	Center-Line Elevation m	Size, m	Maximum Flow Capacity m ³ /sec
1	148.4	1.22 × 2.14	8.49
2	146.4	1.22 × 2.14	8.49
3	141.8	0.92 × 1.22	4.25
4	137.3	0.92 × 1.22	4.25
5	131.2	0.92 × 1.22	4.25

(Continued)

Port No.	Center-Line Elevation m	Size, m	Maximum Flow Capacity m ³ /sec
6	125.1	0.92 × 1.22	4.25
2 floodgates	121.7	1.53 × 1.83	80.10 (both)

This design was selected after numerous options were tried consisting of four or more selective withdrawal ports in two wet wells. Six ports were determined to be the minimum number required for efficient operation. The location of the ports can be varied as much as ± 0.5 m without affecting the operation of the structure.

65. The water supply would be taken out of the bottom of the wet wells. The maximum selective withdrawal capacity would remain the same so that large spring flows could be passed through the two upper ports. This design would provide better control of outflows through the use of more ports and the dual wet well system. It would also save money because the tower could be made narrower and the smaller ports would cost less to fabricate. For these and other reasons, this alternate design was used throughout the remainder of this study.

66. The releases from the alternate design followed the temperature objective (Figure 29) except for a brief period in the spring. If the top port were lowered to the same elevation as the second port (146.4 m), then the release temperatures would have remained low for a longer period (Figure 30). The design was also checked on the other study years and outflow schemes (Figures 31-36). All were able to meet the target objective except for the 58 and 110 Trenton supplements in 1969 (Figure 36). In these two instances, problems occurred whenever the releases were close to the minimum release, $0.17 \text{ m}^3/\text{sec}$. These small flows could be released through only one port. More selective withdrawal ports would be required to match the temperature objective under these conditions. Typical port operations required to meet the release objective are shown in Figure 37.

67. A general regulation scheme to enable the proposed project to meet the downstream temperature objectives (Figure 24) is given in Table 10. This scheme is based on the model results for the years 1969,

1973, and 1974. Variations from year to year are expected. Surface withdrawal is used during the spring to keep the downstream waters warm. During the summer, water is released from progressively lower ports to follow the metalimnion. Bottom withdrawal is used during the winter months to help alleviate possible low DO concentrations. If the lake should go anoxic, problems are not expected downstream. Aeration in the conduit and stilling basin will raise the DO to 85-100 percent saturation.

Trexler predictions

68. Using the coefficients from the Beltzville simulation, the thermal structure of proposed Trexler Lake was simulated for the three study years. In all 3 years, the lake was strongly stratified and followed similar stratification cycles (Figures 38-40). Stratification formed within 2 weeks after the simulation started and lasted through the end of October. Thermocline depths were also similar although surface and hypolimnetic temperatures varied.

69. Zones of inflow for the study years are shown in Figures 41-43. The inflowing water was distributed throughout the water column in early spring and late fall when the lake was isothermal. During periods of stratification, the inflow was constrained to a narrow band within the metalimnion. This phenomenon was caused by small inflows that were cooler than the epilimnion of the lake. In 1974, the inflow did not penetrate as deeply into the lake as in 1973 and 1969. The fall overturn also occurred earlier in 1974.

70. The zones of outflow (Figures 44-46) are similar to the inflow zones except they are much wider as a result of blending between ports. As with the inflow, the withdrawal zones did not penetrate as deeply in 1974, probably due to colder hypolimnetic temperatures.

Water Quality Simulations

71. Twenty-five water quality simulations were made of proposed Trexler Lake to predict its trophic status and determine its sensitivity to various coefficients and update parameters. Although

21 water quality variables were simulated, only temperature, DO, algae, and fecal coliforms are of primary interest and will be discussed. The remaining variables will be used to explain variations in these parameters. The simulation results will be discussed according to base conditions, reservoir outflow regulation schemes, and sensitivity of results.

Base conditions

72. In 1974, both ALGAE 1 and ALGAE 2 responded to the high PO_4 -P initial conditions (Figure 47). Phosphorus concentrations in the surface waters never returned to the initial value of 0.01 mg/l during the simulation cycle. Following the initial bloom, the algae remained at low levels due to phosphorus limitation until day 290 when a second bloom of ALGAE 1 occurred. This bloom resulted from a gradual increase in phosphorus throughout the summer.

73. Algal production occurred deep in the pool near the 1 percent light level where light became limiting. This is clearly illustrated in Figure 48 by the maximum in the DO profile for days 110-170 and in Figure 49 where the algae in the top 1 m are compared with the standing crop in the euphotic zone. This light level was also distinguished by a strong gradient in phosphorus concentrations. Above this level, phosphorus concentrations were 1 μ g/l or less and the inflowing phosphorus was barely sufficient to support the existing standing crop. The maximum production occurred around the 1 percent light level where phosphorus was available from the lower metalimnion and upper hypolimnion.

74. The DO profiles for 1974 (Figure 48) were typical of those found in Beltzville and other lakes. The DO started to go out of the lower metalimnion on day 150. By day 310, the hypolimnion was anoxic. The demand came from algae settling and respiring and detritus accumulating and decaying. Oxygen returned to the lower layers at fall overturn.

75. The wet year, 1973, differed significantly from 1974. Several large blooms of ALGAE 1 and ALGAE 2 occurred before day 190 (Figure 50). Magnitudes of algae standing crops ranged from 1.2 to 2.5 mg/l. These blooms were in direct response to storm events and the

large inflows and nutrient loadings around days 93, 150, and 180 (Figure 20). Carbon was predicted to be limiting during all the blooms but the first one, where phosphorus was limiting. There was also a short period of phosphorus limitation around day 165 that gave ALGAE 2 a competitive advantage over ALGAE 1. This resulted in the bloom of ALGAE 2 on day 175 when sufficient phosphorus became available. After day 190, the ALGAE 1 population fluctuated with the availability of nutrients. As the algae respired or decayed, nutrients were released, regenerated, and became available for re-uptake and growth. Since carbon was limiting during this period, the atmosphere also acted as a source of nutrients.

76. The DO profiles for 1973 were similar to those for 1974 except that the DO went out much faster in the metalimnion and hypolimnion (Figure 51). DO was completely gone by day 210 and did not return until the fall overturn on day 315. The larger oxygen demand in 1973 resulted from detrital input and decay and more algal production.

77. The algae and DO in 1969 followed the same trends as those of 1974 (Figures 52 and 53). An initial bloom of ALGAE 1 and ALGAE 2 occurred in the spring due to high initial conditions of phosphorus. Insufficient phosphorus inflows and phosphorus limitation prevented any more blooms until the middle of October. The algae did not even respond to the storm on day 210 (Figure 19). The DO profiles (Figure 53) showed oxygen demands in the metalimnion and at the sediment-water interface. The hypolimnion went anoxic on day 290.

78. Coliforms were found not to be a problem in any of the study years. All predictions remained below the standard of 200 colonies/100 ml.

Outflow regulations

79. The different outflow regulation schemes provided by NAP were used to determine the sensitivity of the simulations to inflow and outflow. Changes in outflow affected the simulations through the cumulative effects of mixing. This was especially true when the inflow entered the pool at a different level from the outflow. When the total outflow was increased from $1.27 \text{ m}^3/\text{sec}$ to $1.70 \text{ m}^3/\text{sec}$, changes in the

mixing regime were reflected in the temperature profiles for 1974, 1973, and 1969 (Figures 54, 55, and 56, respectively). Differences in both surface temperatures and thermocline depths are apparent. Similar differences occurred when the 1969 outflows for the 110 Trenton supplement were used (Figure 57). All results from the 58 Trenton supplement were indistinguishable from the 110 supplement and will not be discussed further. All statements regarding the 110 supplement are also appropriate for the 58 supplement.

80. In 1974, increasing the outflow to $1.70 \text{ m}^3/\text{sec}$ had little effect on the standing crop of algae (Figure 58). No differences were apparent until day 230. ALGAE 2 dominated through day 290, and the fall bloom of ALGAE 1 on day 300 was one half its value under the base outflow regime. Differences in DO were minimal and related more to changes in temperature structure than algal biomass (Figure 59).

81. Increasing the outflow in 1973 had a more profound effect on algae standing crops (Figure 60). Blooms occurring after day 140 were moved forward 5 to 10 days. The ALGAE 1 bloom on day 150 was also reduced in magnitude. Here the larger outflow and resultant mixing dispersed the phosphorus and algae throughout the euphotic zone, diluting the higher phosphorus concentrations previously available in the metalimnion. The bloom was smaller because less phosphorus was available for growth. As in 1974, the change in the thermal structure had an effect on the DO profiles (Figure 61). With the increased mixing, the fall overturn and return of DO to the lower layers occurred 5 days earlier. The shift in the timing of the blooms caused the DO to be supersaturated on day 170.

82. The $1.70\text{-m}^3/\text{sec}$ total outflow, 110 Trenton supplement, and 58 Trenton supplement regulation schemes were simulated in 1969. No change in algae occurred with the $1.70\text{-m}^3/\text{sec}$ outflow through day 225 (Figure 62). At this time, ALGAE 1 became dominant and remained so through the simulation period. This switch in dominance appeared to be a direct consequence of a storm event on days 210-225. With the $1.70\text{-m}^3/\text{sec}$ outflow, the reservoir operation schedule did not have to be disrupted to release storm event flows, so the outflows remained

constant throughout the simulation period. However, with the base simulation of $1.27 \text{ m}^3/\text{sec}$, the outflow during the period 210-225 had to be increased to $13 \text{ m}^3/\text{sec}$ to maintain the pool elevation below the maximum permissible summer level. The impact of this event was to change the dominance in the phytoplankton.

83. The increased algal biomass between days 230 and 250 resulted in a larger metalimnetic oxygen demand (Figure 63). Otherwise, the DO profiles remained unaffected. ALGAE 1 was also dominant during the last half of the year for the simulations using the outflows for the 58 and 110 Trenton supplements. Here the change in dominance came at a time of minimum release of $0.17 \text{ m}^3/\text{sec}$ rather than at high flow (Figure 64). Again, changes in DO profiles (Figure 65) were attributable primarily to changes in the temperature structure.

84. Surface withdrawal was simulated for 1974 and 1973 to determine the effects of selective withdrawal on water quality. No significant changes were observed from the base simulations. As expected, the epilimnion became shallower as shown for 1973 in Figure 66. In 1973, the second bloom of ALGAE 2 did not occur, and ALGAE 1 dominated after day 140 (Figure 67). The DO profiles reflected the change in thermal structure (Figure 68).

Sensitivity

85. The sensitivity of the model predictions to coefficient and update selections was investigated by perturbing several coefficients and updates from their nominal value. The variables to be perturbed were determined from the results of the algal bioassays and base simulations and are summarized in Table 11.

86. Since the bioassays indicated that phosphorus would be potentially limiting, updates and coefficients pertaining to phosphorus were investigated. Half-saturation coefficients for carbon and light were investigated because carbon and light were found to be limiting at times in the base simulations. Sensitivity results for the light half-saturation coefficient will not be discussed because the model predictions were found to be insensitive to changes in the coefficient.

87. The phosphorus half-saturation coefficient was increased to

investigate phosphorus limitation. With the low phosphorus concentrations in 1974, increasing the half-saturation coefficient resulted in greater phosphorus limitation and consequentially a greater competitive advantage to ALGAE 2. The ALGAE 2 population showed a slight increase while ALGAE 1 showed a slight decrease (Figure 69). The total algal biomass, however, remained similar. DO showed little change over the base simulation (Figure 70). The shifting of the last bloom resulted in the maximum in the DO profile on day 310.

88. Perturbing the carbon half-saturation coefficient in the 1974 simulation had little effect because phosphorus was limiting. However, in the 1973 simulation, when carbon was limiting, a decrease in the carbon half-saturation coefficient resulted in larger blooms (Figure 71). A smaller half-saturation coefficient meant that more growth could occur at smaller concentrations of the limiting nutrient. The effect of the larger blooms on DO was masked because DO was already depleted from the hypolimnion (Figure 72). Changes in the timing and magnitude of blooms did, however, result in increased DO in the surface waters on day 290.

89. Decreasing the settling rates of ALGAE 1 and 2 increased the initial blooms of ALGAE 1 and 2 (Figure 73). The bloom of ALGAE 1 on day 150 never materialized. Instead the ALGAE 2 bloom occurring on day 175 was moved forward about 15 days. The decay of the larger initial blooms resulted in more respiration and carbon dioxide (CO_2) production with a concomitant decrease in pH. On day 145, the algae were therefore phosphorus limited instead of carbon limited as in the base simulation. Since ALGAE 2 had the advantage under phosphorus-limited conditions with its smaller phosphorus half-saturation coefficient, it naturally replaced the ALGAE 1 bloom. Once carbon limitation returned on day 180, ALGAE 1 regained the competitive advantage and dominated for the remainder of the simulation. The DO profiles reflected the changes in the phasing of the algae blooms (Figure 74). For instance, the peak in DO at the 28-m elevation on day 150 disappeared since bloom conditions no longer existed at this time. Similarly, since day 170 fell within a bloom instead of between blooms, the peak in the DO profile denoted algal production.

90. Increasing the settling rates did not have as large an effect on the plankton assemblages. The initial blooms of ALGAE 1 and 2 decreased slightly (Figure 75). The subsequent blooms moved forward about 5 days. After day 210, the results were similar to the base simulation. Again, the shift in the timing of the blooms resulted in differences in the surface concentrations of DO (e.g., day 170 in Figure 76). The hypolimnion went anoxic sooner because the larger settling rates increased the concentration of algae in the aphotic zone, thereby increasing respiration and decay.

91. Phosphorus inflows were doubled and halved in the three study years to investigate the sensitivity of the simulations to phosphorus loadings. In 1974, doubling the phosphorus inflows increased the algae concentrations through day 130 (Figure 77). Thereafter, the algae concentrations remained similar to the base conditions until day 235 when ALGAE 1 became dominant. Since ALGAE 1 had a greater growth rate, the increased phosphorus loadings gave ALGAE 1 a competitive advantage over ALGAE 2 even though ALGAE 2 had a lower phosphorus half-saturation coefficient. The DO profiles reflected the increased algae with a larger metalimnetic and hypolimnetic oxygen demand (Figure 78). Therefore, DO went out in the hypolimnion earlier in the year. Halving the phosphorus input had little effect because the base simulation was already severely phosphorus limited (Figure 79). The major difference was a reduction in the ALGAE 1 bloom on day 300. The DO was slightly better at times (e.g., day 230), but it generally was indistinguishable from the base simulation (Figure 80).

92. Doubling the phosphorus inflows in 1973 increased the ALGAE 1 bloom on day 110 to 3.75 mg/l dry weight (Figure 81). The simultaneous ALGAE 2 bloom was decreased slightly. The remainder of the year was relatively unaffected because the algae were carbon limited. The DO profiles showed a slight decrease in oxygen in the hypolimnion after day 130 but no significant differences (Figure 82). The increase was due to the increased demand created by the larger ALGAE 1 bloom settling and decaying. Reducing the phosphorus inflow by one half halved the first ALGAE 1 bloom but had little effect on the ALGAE 2

bloom (Figure 83). After day 140, ALGAE 1 dominated completely. Carbon limitation on days 160 through 165 gave ALGAE 1 the advantage. ALGAE 2 needed a period of phosphorus limitation to overtake ALGAE 1 in the base simulation. Larger blooms on days 195 and 225 occurred because there was sufficient CO_2 available for growth. The DO improved slightly, but the hypolimnion still went anoxic by day 210 (Figure 84).

93. 1973 was also simulated by using a constant phosphorus input of 0.018 mg/l, the mean annual average. The algae concentrations were greatly reduced, especially during the spring when phosphorus was limiting (Figure 85). In addition, ALGAE 2 dominated over ALGAE 1 until day 140. After day 170, carbon was again limiting. The DO concentrations in the hypolimnion were improved, but anoxic conditions still prevailed.

94. Doubling the phosphorus in 1969 yielded results similar to the constant phosphorus input in 1973 (Figure 86). ALGAE 2 bloomed first; but by day 145, ALGAE 1 completely dominated the simulation. After day 250, the population steadily increased. This resulted from a unique series of events. Phosphorus was limiting until day 235, when carbon became limiting. At this time phosphorus was being supplied faster than it could be utilized, and it therefore started to accumulate. Days 290-300 were characterized by low light levels, and the algae were light limited throughout the entire water column. This gave the CO_2 an opportunity to be replenished through reaeration and respiration. When the light returned to higher levels, there were sufficient nutrients available for the large bloom. As expected, the increased algae resulted in more DO depletion in the metalimnion and hypolimnion (Figure 87). Halving the phosphorus resulted in some minor differences in algae concentration after day 220 (Figure 88). Changes in DO were minimal.

95. Since no fecal coliform violations were encountered in the base simulations, fecal coliform inflows were increased by an order of magnitude. No violations were encountered in 1969, but some did occur in 1973 and 1974. In 1974, fecal coliforms exceeded the maximum Pennsylvania standard of 200 colonies/100 ml for about 5 days at the

beginning of the simulation (Figure 89). A similar period of violation lasted about 15 days in 1973 (Figure 90). In addition, two storm events in 1973 resulted in smaller zones of violation during the summer.

Monte Carlo simulations

96. There were insufficient data for accurate definition of required coefficients and updates for this study. Many of the coefficients also vary with time in a manner that is currently beyond mathematical description. One way to estimate the uncertainty in the predictions resulting from these and other inadequacies is to vary a coefficient and observe the perturbations. This was done in the preceding section on sensitivity analyses. A more rigorous approach is to use Monte Carlo simulations.

97. In Monte Carlo simulations, any number of coefficients and updates can be varied simultaneously within prescribed limits and distributions. Specifically, new values for the coefficients and updates are selected at random from the specified distributions during each computational step. Every simulation will therefore be different. Several of these simulations are then superimposed on one another to define the limits of the predictions.

98. Two types of Monte Carlo simulations were used in this study. Coefficients were varied in the first set of simulations, and updates were varied in the second set. Some parameters, such as mass fractions and stoichiometric relationships, were always held constant to prevent mass imbalances from occurring. The coefficients and updates that were varied are summarized in Tables 12 and 13.

99. Coefficient distributions were determined from data published in the literature (Appendix D). When there were not sufficient data to define a distribution, a uniform distribution was assumed. Limits were established to include all known data. Uniform distributions were assumed for all decay rates, all settling rates, and carbon and light half-saturation coefficients. The distribution of growth rates for freshwater species expected to occur in Trexler Lake was found to be normal (Figures 91 and 92). This was also found to be true for all respiration rates and zooplankton growth and mortality rates.

100. The distributions for phosphorus and nitrogen half-saturation coefficients were found to be skewed to the left (Figure 93). They resembled a rotated log normal distribution. One reason for this skewness could be the ambiguous definition of a eutrophic lake. Since only coefficients from eutrophic lakes were considered and since the half-saturation coefficients increase with nutrient concentration (Hendrey and Welch 1973; Carpenter and Guillard 1971; and Toetz et al. 1973), any eutrophic lake characterized by low nutrient concentrations would also be characterized by low half-saturation coefficients. However, a lake considered eutrophic in one part of the country may not be considered eutrophic in another part. The inclusion of coefficients from these lakes would tend to skew the distribution to the left. Another reason for the sharp decline at large concentrations could be that another nutrient is limiting.

101. The scatter plots of the updates with flow (Figures 2, 3, 5-8, 12) indicated very little, if any, relationship with flow. Updates of phosphorus, ammonia, fecal coliforms, and detritus were therefore assumed to be uniformly distributed. Nitrate was not included in the analyses because its concentrations were so large compared with the others that changing it would have little impact.

102. Twenty iterations were run in groups of ten, with coefficient variation and with update variations on the 1974 data set. The coefficient variation resulted in ALGAE 2 dominating throughout the simulation (Figures 94 and 95). This was not observed in any of the previous simulations. Since ALGAE 2 had the advantage over ALGAE 1 only under low temperatures and low phosphorus concentrations, phosphorus limitation must have been important in 1974. There was little variation in the ALGAE 2 concentrations through day 240. The blooms after day 240 were sensitive to coefficient selection.

103. Typical DO concentrations for the hypolimnion, metalimnion, and epilimnion are shown in Figures 96-98, respectively. In the hypolimnion, the DO decreased at an approximately constant rate until it went anoxic on day 290. It remained anoxic for about 30 days. The DO in the metalimnion showed more variation from the mean. The large,

abrupt fluctuations resulted from algal production. This layer was characterized by large phosphorus gradients and sufficient light to permit maximum algal growth. The DO in the epilimnion varied inversely with temperature. The large variations after day 270 coincided with similar variations in algae.

104. The algal blooms predicted from update variation followed trends similar to the base simulations (Figure 99). ALGAE 1 grew immediately and persisted through the summer. A large bloom always occurred after day 270. There appeared to be two distinct times for the bloom to begin: one around day 290 and the other on day 300. ALGAE 2 bloomed for a 30-day period in the spring and then gradually decreased (Figure 100).

105. In the update variation simulations, the detritus (i.e., organic loading) input was varied. This resulted in higher organic loadings as evidenced by the lower DO in the hypolimnion (Figure 101). It went anoxic sooner and stayed anoxic longer than during the coefficient variation simulation. The metalimnion was also characterized by a larger oxygen demand and lower DO (Figure 102). From day 90 to day 230, there was a constant decline in DO. In the epilimnion, the results were similar to the results from coefficient variation (Figure 103).

Discussion

106. Before the mathematical simulation results are discussed, it is important to realize that ecosystem modeling is as much an art as a science. No model is capable of predicting absolute values. When ecological systems are simplified to a few compartments and coefficients, as is the case with any model, mean coefficients may result in predictions that are off by as much as an order of magnitude from measured values. A broad range of coefficients is necessary to define all possible perturbations of the system. Monte Carlo simulations and sensitivity analyses are two ways to estimate the extent of these perturbations. In this study, the predictions were found to be

relatively insensitive to variations in coefficients and updates.

107. Consideration must also be given to the model assumptions and limitations (paragraph 31), the most important one being the one-dimensional assumption. The predictions are valid only in the deep part of the pool near the dam, not in the headwaters, coves, or embayments. The predictions are also valid only under aerobic conditions. It may be possible to predict when the DO goes out, but there is no mechanism in the model to account for the oxygen debt that builds up under anoxic conditions. Lastly, model predictions represent conditions after the transients in water quality from the initial filling have diminished. This may take 5 years or more.

Thermal simulations

108. The mathematical simulations indicated that proposed Trexler Lake would stratify early in April and remain strongly stratified until fall overturn in mid-November. The downstream temperature objective was met using a selective withdrawal structure with six ports in two wet wells. This allowed for controlled blending between ports in different wet wells. Blending between ports in the same wet well is difficult to control under stratified conditions because of density blockage. Six ports are necessary because of the depth of proposed Trexler Lake.

Water quality simulations

109. Algae. Predicted algal blooms ranged from 1.0 to 2.5 mg/l. For comparative purposes, 0.7 mg/l is considered a visible bloom and 1.5 mg/l is considered a nuisance bloom. If the conversion factor developed by Spangler (1969) is used (i.e., 0.23 mg/l dry weight = 1 mg/m³ chlorophyll a), then the magnitude of the predicted algal blooms in terms of chlorophyll a would be 5-11 mg/m³. This is the same magnitude as found in the surrounding lakes (Table 7).

110. The predicted algae concentrations did not show any set pattern of succession. Simple changes in coefficients, outflows, or inflows resulted in changing patterns of dominance between the two types of algae or changes in the phasing of the blooms. There was, however, a tendency for diatoms (ALGAE 2) to dominate in dry or average years (1969 and 1974), characterized by low phosphorus loadings, and

for green and blue-green algae (ALGAE 1) to dominate in wet years (1973), characterized by phosphorus loadings.

111. Doubling the phosphorus loadings increased the magnitude of the blooms, but halving the loadings did not result in large reductions. This would tend to indicate that improving the effluents from the Heidelberg Heights sewage treatment plant would have little effect on the overall quality of the lake. It would, however, probably improve local conditions in the headwaters where Mill Creek enters the impoundment. This area cannot be simulated by the model.

112. In some of the simulations, diatoms dominated through most of or all of the year (e.g., Figures 9⁴ and 95). These predictions may not be completely valid because diatoms require silica for growth and the model does not consider silica. If silica should become limiting and there is sufficient phosphorus available for growth, then it is highly probable that the green and blue-green algae will dominate, provided other conditions are favorable. Lund et al. (1963) found that *Asterionella* required 0.5 mg/l SiO_2 for growth in Lake Windermere. No silica data were found for Jordan Creek, but concentrations in Francis E. Walter Lake were 1.0 to 1.5 mg/l. If similar concentrations occur in proposed Trexler Lake, then there should be sufficient silica available for diatom growth.

113. Dissolved oxygen. Predicted DO profiles were similar to those measured in Beltzville Lake (e.g., Figures 16 and 17). Oxygen demands were evident in both the metalimnion and hypolimnion. The model predicted that the hypolimnion would go anoxic at the end of September in 1969 and 197⁴ and at the end of July in 1973. These dates are earlier than observed in Beltzville Lake. This could reflect higher organic loadings to proposed Trexler Lake. Oxygen did not return to the lower layers of proposed Trexler Lake until fall overturn.

114. Because the model cannot simulate anoxic conditions, no conclusions can be drawn concerning the possibility for hydrogen sulfide production or the release of iron, manganese, and nutrients under anoxic conditions. Since these problems do not appear to be serious in any of

the surrounding lakes that go anoxic, there may not be a problem with them in proposed Trexler Lake.

115. Fecal coliforms. Simulated fecal coliforms did not exceed the state standard of 200 colonies/100 ml. It was only after the inflow concentrations were increased by a factor of ten that the simulations showed violations of state standards. During these periods, the inflowing counts were approximately 10,000/100 ml or the same order as the maximum value measured (Table 2). Some violation of the standards will probably occur in the headwaters because the inflows are not instantaneously dispersed throughout the pool as assumed by the model.

116. pH. The model also predicted that the hypolimnion would become slightly acidic (~6.8 or lower). This prediction is consistent with field measurements in surrounding impoundments.

PART V: LOADING ANALYSES

117. Cultural eutrophication is generally accepted as an acceleration of the natural eutrophication process due to man's activities in the watershed. The activities result in excess nutrient discharges into surface waters. These nutrient discharges occur from both point (e.g., sewage and industrial outfalls) and nonpoint (agricultural and urban runoff) sources in the watershed. One of the earliest studies attempting to quantify the trophic status of lentic environments based on external nutrient loads was conducted by Vollenweider (1968). Since 1968, numerous studies have been conducted to modify and improve the nutrient loading concept (Shannon and Brezonik 1972; Dillon 1974, 1975; Dillon and Rigler 1975; Kirchner and Dillon 1975; Larsen and Mercier 1976; Vollenweider 1975, 1976; Carlson 1977). The use of nutrient loading models also has been extended to predict water clarity and average chlorophyll concentrations (Carlson 1977; Dillon and Rigler 1974; Vollenweider 1976; Jones and Bachmann 1976).

118. Since the nutrient loading concept is predicated primarily on the external nutrient load and various morphometric characteristics of the impoundment rather than measured in-lake quantities, it may be applied for predictions of the trophic status expected to occur in proposed impoundments. Several of the nutrient loading models, therefore, were applied to project the trophic status in the proposed Trexler Lake.

Assumptions and Limitations

119. It is important to consider the assumptions of the nutrient loading models in assessing the trophic status of an impoundment before any definitive conclusions can be drawn. While the equations are simple and relatively easy to apply, it should not be inferred that conclusions concerning the trophic status can be directly and simply drawn from the results. The relative position of a proposed or existing impoundment on a loading plot, specifically in the eutrophic

zone, does not necessarily indicate the magnitude or degree of eutrophication. While the nutrient loading concept is a reasonable approach for demarcation of the general trophic status, e.g., oligotrophic, mesotrophic, or eutrophic, there are many other factors that affect the trophic state of a reservoir such as the zone of withdrawal, stratification, alkalinity, cation ratios, and others.

120. Several assumptions have previously been stated by Dillon (1974):

- a. Loading, flushing, and sedimentation rates are assumed to be in steady state.
- b. Stratification or nonuniform mixing is ignored; the system is considered to be a well-mixed reactor.
- c. The concentration of a substance in the outflow is equated to the mean concentration in the lake.

121. Several corollary assumptions follow:

- a. The substance under consideration is limiting.
- b. There is a direct relation between loadings and biomass.
- c. A reduction in loadings will result in a concomitant reduction in indicators of eutrophication.

122. While the applicability and validity of the assumptions will be discussed later, it is important to refer to the assumptions during the subsequent calculations of trophic status for proposed Trexler Lake.

Application of Nutrient Loading Models

123. Water quality samples collected at the proposed Trexler damsite by the USGS from 1972 to 1975 were used to develop a regression equation of total phosphorus versus flow. This was used to generate an annual phosphorus load to the proposed project based on daily flow records. Three regression equations (linear, quadratic, and exponential) were tested. The linear equation

$$\text{Total P} = 0.004 + 0.014Q$$

where Q = flow in m^3/sec , had the highest coefficient of determination ($R^2 = 0.58$) or accounted for the greatest portion of the variance. Flows for the three study years (1969, 1973, and 1974) were used to generate phosphorus loads representative of dry, wet, and average years, respectively. External phosphorus loading values were computed using the equation of Vollenweider (1975) and are plotted in Figure 104. The equations of Dillon and Rigler (1974), Larsen and Mercier (1976), and Vollenweider (1976), which account for sedimentation, were used to compute average in-lake phosphorus concentrations. These equations and the average phosphorus concentrations are shown in Table 14. The retention coefficients and their respective equations are shown in Table 15. The loading values calculated using the Vollenweider (1976) equation are plotted in Figure 105.

124. Carlson (1977) developed a trophic state index (TSI) that interrelates commonly accepted criteria of eutrophication such as Secchi disk depth, chlorophyll a concentrations, and phosphorus concentrations if phosphorus is the limiting nutrient. Measurement of any one of these variables may be used to compute a numerical value that represents the trophic state of the system and may also be used to compute estimates of the other two variables. TSI values computed from estimates of average in-lake phosphorus concentrations are listed in the following tabulation:

<u>P Equation Source</u>	<u>TSI (Carlson 1977)</u>		
	<u>1969</u>	<u>1973</u>	<u>1974</u>
Dillon and Rigler (1974)	54.0	63.2	55.8
Larsen and Mercier (1976)	55.4	64.8	57.3
Vollenweider (1976)	55.4	64.8	57.0

These average phosphorus values were also used to obtain estimates of the chlorophyll a concentrations expected in the proposed Trexler Lake. Chlorophyll a is a surrogate variable used to represent algae since chlorophyll concentration is easier and less time-consuming to obtain than algae counts or weights. The equations of Dillon and Rigler (1974), Vollenweider (1976), and Carlson (1977) and predicted

chlorophyll concentrations are listed in Table 16.

125. Regression equations for total phosphorus concentration versus flow also were developed for Upper Jordan, Switzer, Mill, and Lyon creeks. Three equations (linear, quadratic, and exponential) were tested, and the equational form with the highest value of R^2 was retained. The equations and R^2 values for each creek are shown in the following tabulation:

<u>Creek</u>	<u>Regression Equation</u>	<u>R^2</u>
Upper Jordan	$P = 0.008 + 0.039Q$	0.40
Switzer	$P = 0.018 + 0.053Q$	0.39
Mill	$P = 0.069Q^{-0.253}$	0.17
Lyon	$P = 0.018 + 0.086Q$	0.33

Note: P = total phosphorus concentration, g/m^3

Q = flow, m^3/sec

126. Continuous flow records were not available for any of the tributaries. Flow measurements were obtained only at the time of water quality sampling, which generally was conducted monthly. To obtain a relative estimate of the total phosphorus loadings from each tributary, simulated average daily flows for the period of April-October from 1966 to 1975 were obtained from the Hydrocomp Simulation Program (HSP) (Appendix B) and used to generate phosphorus concentrations for this period. Average flows and phosphorus concentrations during the months of January-March and November and December for all study years were used to obtain average monthly loading estimates for each of these five months. The winter months were not calibrated in the HSP model since snow and evaporation data were not available for the watershed at the time of simulation. The purpose of these calculations was to obtain the relative contribution of each tributary to the overall phosphorus loading in the proposed project. Annual phosphorus loading estimates by tributary are shown in Figure 106.

127. The nutrient loading models, Carlson TSI, and predicted chlorophyll concentrations all indicate that the proposed Trexler Lake

will be eutrophic. The critical external phosphorus loadings for 1969, 1973, and 1974, based on permissible in-lake phosphorus concentrations of 10-20 $\mu\text{g}/\ell$ (Vollenweider 1976), were computed using the Vollenweider equation and are compared with the predicted external loadings to the proposed Trexler Lake in the following tabulation. The critical phosphorus level delineates the transition range between oligotrophic and eutrophic systems.

	Predicted External Loading, $\text{g P}/(\text{m}^2 \cdot \text{yr})$		
	<u>1969</u>	<u>1973</u>	<u>1974</u>
Permissible in-lake concentration:			
10 $\mu\text{g}/\ell$ P	0.15	0.33	0.28
20 $\mu\text{g}/\ell$ P	0.30	0.67	0.55
P-Flow Regression Equations	0.74	2.23	1.10

128. The permissible in-lake phosphorus concentrations (10-20 mg/m^3) result in TSI values of 38-47. A comparison of these values with the values listed in the tabulation in paragraph 124 also indicates a eutrophic system. In addition, the predicted chlorophyll a concentrations are greater than the 10- mg/m^3 chlorophyll concentrations used to distinguish mesotrophic and eutrophic systems (Gakstatter et al. 1975).

129. While the phosphorus loadings from each tributary are generally less than the critical loading computations, their combined input would exceed the critical values (Figure 106). The regression equations indicate that the phosphorus loadings from Upper Jordan, Switzer, and Lyon creeks increase with flow while the loadings from Mill Creek generally decrease with flow (paragraph 125).

Discussion

130. Based on loading analyses, proposed Trexler Lake is expected to be a eutrophic impoundment. This finding should be accepted with caution for two reasons. First, all the phosphorus data were generated

and therefore subject to error. Second, it is questionable whether a reservoir such as proposed Trexler Lake meets all of the assumptions (paragraph 119).

131. Under some circumstances, these assumptions are violated. For example, storm events and the loadings associated with the hydrograph or elevated flow represent unsteady loadings; yet, most of the loadings to impoundments occur during storm events. Through judicious operation of a selective withdrawal structure, the high nutrient and suspended solids loads can be passed through the impoundment, thereby increasing the flushing rate and decreasing retention. This, however, violates the assumption of a steady-state system.

132. Model simulations and comparisons with surrounding impoundments indicate that the proposed Trexler Lake will stratify, thereby violating the assumption of a well-mixed system. Inflowing nutrients will not be uniformly mixed throughout the pool. In general, there is an exponential decay in nutrient concentrations from the headwaters to the dam (Ott et al. 1973). Stratification and the operation of the selective withdrawal structure will probably result in outflow concentrations that are different from average in-lake concentrations. This obfuscates the assumption of equivalent concentrations in the outflow and in the lake water column.

133. Greater concentrations of phosphorus should be released from the hypolimnetic ports than from the epilimnetic ports during stratification (Dunst 1974; Moore 1976). During periods of stratification, inflowing stream waters that enter as an interflow or underflow can be withdrawn through metalimnetic or hypolimnetic ports so that the associated nutrient load does not become available to epilimnetic phytoplankton. Dunst (1974) found that significantly greater amounts of phosphorus were released in hypolimnetic discharges than epilimnetic discharges during stratification. This increase was a function of two factors. First, the inflowing stream entered as an underflow and passed directly through the hypolimnion, being released out of the bottom gates. Secondly, the hypolimnion became anoxic and phosphorus was resolubilized, released into the overlying waters, and discharged through the bottom

ports. Hypolimnetic withdrawal, in this system, served to purge the system of phosphorus. This differential release of phosphorus from the stratified system as well as the interflowing and underflowing stream waters may enhance water quality in the proposed project by reducing the phosphorus available for phytoplankton uptake during stratification and for redistribution at overturn.

134. Algal bioassays and nutrient ratios both indicate that phosphorus will probably be the limiting nutrient in the proposed Trexler Lake. Since phosphorus is the substance under consideration, this assumption is met. Several authors (Dillon 1974; Jones and Bachmann 1976; Vollenweider 1976) have developed relations between chlorophyll and phosphorus, leading to the assumption that there is a relation between loadings and biomass. This assumption is tempered, however, by the requirement of a direct relation. A direct relation would indicate that a reduction in loadings would produce a concomitant reduction in biomass. A reduction in phosphorus, however, will not necessarily result in a decrease in chlorophyll. Emery et al. (1973) found that diversion of sewage from Lake Sammamish, Washington, had little impact on its trophic state. An 85 percent reduction in loadings to several Swedish lakes was required before any improvement in water quality was detected (Forsberg et al. 1975).

135. The chlorophyll predictions for 1969 and 1974 (Table 16) are within the range found in surrounding impoundments (Table 7), but the values for 1973 are approximately twice as large as those measured in these surrounding lakes. While the predictions reflect the increased flow during 1973, reservoir operation may permit the discharge of these elevated flows more rapidly than would occur in natural lakes. This would reduce the effective loadings to the impoundment and would possibly diminish the chlorophyll concentrations expected in the proposed Trexler Lake.

136. As indicated earlier, the total phosphorus values used in the nutrient loading calculations were generated from a regression equation. The regression equation used to predict phosphorus concentrations is based on flow and will, therefore, include elevated or unsteady

flow events. This will result in higher loading values than would occur under steady flow conditions since steady flow usually represents base flow. While these calculations probably result in more representative total phosphorus loadings to the proposed impoundment, the inclusion of elevated flows may artificially raise the position of the proposed project on the loading plots since the loading plots are based on a steady-state system. There will be some compensation, however, since the areal water loading or flushing rates will also incorporate these elevated flows. The extent of this compensation, however, is unknown.

137. The regression equations for the individual tributaries indicated that increased flow resulted in a dilution effect on Mill Creek loadings and increased loadings from the other tributaries. This relationship was expected since the effluent from the Heidelberg Heights sewage treatment plant produced high phosphorus concentrations in the creek at low flow. During higher flows, the phosphorus load was distributed through a greater volume, resulting in decreased concentrations. The relative loading values also indicated that advanced treatment of the sewage effluent will probably not decrease the overall eutrophic state of the reservoir although it would improve local conditions in the Mill Creek arm of the proposed Trexler Lake.

138. Considering the violations of the nutrient loading model assumptions, the inclusion of elevated flow events in the total phosphorus regression equation, model simulations, and comparisons with surrounding impoundments, it appears that the nutrient loading models were liberal in the prediction of proposed Trexler Lake's eutrophication potential. A reduction in the total phosphorus loading values would place it in the early stages of eutrophication or in the transition between a mesotrophic and a eutrophic system. This classification would be consistent with the trophic status of surrounding impoundments.

PART VI: DISCUSSION OF PREDICTIONS AND CRITERIA

139. In the studies described in the preceding sections, different techniques were used to predict the water quality and trophic status of proposed Trexler Lake. The predictions will be discussed, compared, and related to appropriate water quality criteria in this section. Trophic state, algae, DO, fecal coliforms, pH, nitrate, Heidelberg Heights sewage treatment plant, proposed release schedule, and pesticides and heavy metals are discussed.

Trophic State

140. The trophic state of an impoundment refers to the degree of nutrient enrichment. Lakes are usually classified as oligotrophic, mesotrophic, or eutrophic in the order of increasing enrichment. The problem with this classification is that it is subjective, and definitions vary from one part of the country to another.

141. The Great Lakes Group (1976) recommended that concentrations of 7-8 mg/m³ of chlorophyll a separate mesotrophic from eutrophic lakes, while the National Eutrophication Survey (Gakstatter et al. 1975) recommended 10 mg/m³. Using these criteria, the modeling predictions and the data from surrounding impoundments indicate that proposed Trexler Lake would be meso-eutrophic. In contrast, the nutrient loading models predict that it would be eutrophic.

142. It was concluded that proposed Trexler Lake would be meso-eutrophic. The findings of the nutrient loading models were questioned for two reasons. First, all the phosphorus data were generated data and therefore subject to error. Second, it is questionable whether a reservoir such as proposed Trexler Lake meets all of the assumptions of the nutrient loading models (paragraph 119).

Algae

143. The types of algae found in the surrounding impoundments and

expected to occur in proposed Trexler Lake are summarized in Table 6. No problems are expected with any of these except possibly *Ceratium*, which in sufficient numbers could produce taste and odor problems for water supply. The mathematical model was not capable of simulating *Ceratium*.

144. No definite pattern of algal succession was found in the surrounding impoundments. This observation was confirmed by the mathematical simulation. Blooms of diatoms and green and blue-green algae are possible whenever local conditions become conducive for growth.

145. The algal concentrations found in surrounding impoundments ranged from 1.9 to 13 mg/m³ of chlorophyll a (Table 7). This is consistent with the mathematical model predictions of 5 to 11 mg/m³ of chlorophyll a for blooms. Chlorophyll a predictions extrapolated from nutrient loading analyses were higher, possibly for the reasons given in paragraph 142. Larger concentrations of algae are expected in the headwater regions, especially Mill Creek, because of higher nutrient concentrations (Ott et al. 1973). The mathematical model is not capable of simulating this effect because of the one-dimensional assumption.

Dissolved Oxygen

146. The Pennsylvania standard for DO in lakes, ponds, and impoundments is no value less than 5 mg/l at any point. All of the surrounding impoundments went anoxic in the hypolimnion and violated this standard. Proposed Trexler Lake is also expected to go anoxic in the hypolimnion and violate state standards. The mathematical model predicted about one month of anoxia. This is similar to the period measured in Beltzville Lake. No problems are expected under anoxic conditions with downstream releases because aeration in the conduit should increase the DO to 85-100 percent saturation.

Fecal Coliforms

147. The Pennsylvania standard for fecal coliforms is

200 colonies/100 ml based on the geometric mean of five consecutive samples. The EPA recommended criteria for body contact recreation is 200 colonies/100 ml based on a logarithmic mean of a minimum of five samples in 30 days. The value of 200 colonies/100 ml will be used here.

148. The surrounding lakes appear to have no problems meeting these criteria. The model simulations also predicted no problems. However, since the model is one-dimensional and not able to simulate longitudinal variations and since some of the inflow counts were over 200 colonies/100 ml (Tables 1-3), it is expected that periodic violations may occur in the headwater regions. The recreation sites should be located to avoid these areas.

pH

149. The Pennsylvania standard for proposed Trexler Lake for pH is greater than 6.0 and less than 8.5. The EPA recommended pH criterion for freshwater aquatic life is 6.5 to 9.0.

150. In Table 2, pH values in the inflows to proposed Trexler Lake ranged from 6.3 to 9.2. One value in Table 1 was 5.0. The lower pH values occurred during periods of high runoff and were attributed to slightly acidic soils found in the watershed. During periods of low flow, biological activity depleted the free CO₂, thereby elevating the pH to 9.0 and above. This phenomenon also occurs in lakes and is expected to occur in the surface waters of proposed Trexler Lake. Violations may occur, but they will be transient in nature and are not expected to increase water treatment costs.

151. Because of the slightly acidic runoff, the hypolimnion of proposed Trexler Lake is expected to be slightly acidic. Model simulations and data from surrounding CE impoundments confirmed this conclusion. No problems are anticipated here either.

Nitrate

152. Nitrate was considered because water supply was one of the

project purposes. The EPA recommended criterion for nitrate for water supply is 10 mg/l as N. Although nitrate inflows to proposed Trexler Lake are expected to be high, most values should be below 5 mg/l. Both the mathematical simulations and data from the surrounding lakes indicated that nitrate will not accumulate in the lake. No problems are expected because the nitrate concentrations found in the proposed Trexler Lake are expected to be less than in the inflows and because nitrate is not expected to accumulate in the lake.

Heidelberg Heights Sewage Treatment Plant

153. Any time a sewage treatment plant is located upstream of an impoundment, there is concern that the effluents may accelerate the eutrophication process. Higher nutrient concentrations were found in Mill Creek (Table 3), and the algal bioassays confirmed the potential for accelerated eutrophication. The problem is expected to be worse under low-flow conditions because increased flows dilute the effluents. This phenomenon is illustrated by the regression equation developed for total P (paragraph 125). Concentrations of phosphorus in Mill Creek were found to decrease exponentially with increases in flow, while in the other tributaries the phosphorus concentrations increased with flow.

154. Model simulations and nutrient loading analyses indicated that improving the effluents from the Heidelberg Heights sewage treatment plant would have little effect on the overall trophic status of proposed Trexler Lake. Even though nutrient concentrations were higher in Mill Creek, the contribution of Mill Creek to the total nutrient load to the reservoir (i.e. flow \times concentration) was small because of the small flows in Mill Creek (Table 3). Improving the effluents of the Heidelberg Heights sewage treatment plant would, however, improve local conditions in the Mill Creek branch of proposed Trexler Lake.

Proposed Release Schedule

155. The mathematical simulations indicated that proposed Trexler

Lake could be operated to meet the downstream temperature objective provided the selective withdrawal structure is modified to include six selective withdrawal ports in two wet wells. Selective withdrawal ports at six different elevations are needed because of the depth of proposed Trexler Lake. Two wet wells are required to prevent density blocking when flows between two ports are blended under stratified conditions.

156. A typical release schedule to meet the temperature objective would consist of withdrawing water from the top ports during the spring and then gradually stepping down to follow the lower metalimnion (Table 10). Consideration must also be given to operating the project to achieve water supply benefits. Potential problems could arise with iron and manganese if water is taken from an anoxic hypolimnion or metalimnion, and with taste and odor if large numbers of *Ceratium* or other taste- and odor-causing algae are present. If the project is constructed and if these problems do occur, then the water supply benefits may still be realized through judicious operation of the selective withdrawal structure. This may, however, involve conflicting objectives. For example, near-surface withdrawal may be required to avoid iron and manganese problems while hypolimnetic withdrawal may be required to meet the downstream temperature objective. Conflicts of this type can only be resolved at the time they occur, if they do occur.

Pesticides and Heavy Metals

157. Pesticides and heavy metals were not considered in this study. Kaeufer (1973) and Usinowicz et al. (1977) measured concentrations below currently accepted standards. Based on these limited data, no problem is anticipated with either heavy metals or pesticides.

PART VII: CONCLUSIONS

158. Conclusions based on this study are as follows:

- a. Proposed Trexler Lake will probably be mesotrophic or in the early eutrophic phase.
- b. Blooms of diatoms and green and blue-green algae are possible throughout the growing season whenever local conditions become conducive. The magnitude of the blooms will be similar to those of surrounding lakes.
- c. The hypolimnion will go anoxic for about one month prior to fall turnover. During this period, state standards for DO will be violated and problems with iron and manganese release may occur.
- d. There will probably be no problem with fecal coliforms near the dam, but state standards will be exceeded in the headwater regions during periods of high runoff. Recreation sites should be selected to avoid these areas.
- e. The hypolimnion may be slightly acidic.
- f. Proposed Trexler Lake will probably be phosphorus limited.
- g. Nitrate concentrations within the pool will be less than the EPA recommended criterion of 10 mg/l as N.
- h. Improving the quality of the effluent of the Heidelberg Heights sewage treatment plant will improve local conditions in Mill Creek but probably would not change the overall trophic status of proposed Trexler Lake.
- i. The downstream temperature objective can be met provided the selective withdrawal structure is redesigned with a minimum of six ports and with the capability to blend between ports.
- j. Water supply standards will be met, even under low-flow conditions.

159. These conclusions are in general agreement with the comments of Hull (1976) and Sawyer (personal communication*).

* Personal communication from Dr. Clair N. Sawyer, Sun City, Arizona, to Metcalf and Eddy, Inc., concerning the U. S. Environmental Protection Agency's report entitled "A Water Quality Investigation of the Jordan Creek Watershed in the Area to Be Impounded by Trexler Dam," dated 3 November 1976. Dr. Sawyer was Director of Research for Metcalf and Eddy, Inc., Boston, Massachusetts, from 1958 until his retirement in 1971.

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Table 1
Summary of All Water Quality Data Taken at Trexler Project

Variable	Sample Size	Range	Median	Mean
Alkalinity, mg/l	205	13-420	32.0	40.3
Total organic carbon (TOC), mg/l	187	0-34.0	3.0	3.89
Total coliforms, colonies/100 ml	313	6-300,000	2400.0	7610.0
Fecal coliforms, colonies/100 ml	335	0-35,000	505.0	2090.0
NO ₃ -N, mg/l	278	0-11.8	3.40	3.6
NO ₂ -N, mg/l	57	0.002-0.035	0.01	0.0128
NH ₄ -N, mg/l	262	0-5.2	0.050	0.165
Total nitrogen (N), mg/l	31	1.25-6.20	3.2	3.3
Total Kjeldahl nitrogen (TKN), mg/l	243	0.04-5.5	0.56	0.673
Organic nitrogen (N), mg/l	201	0.04-2.5	0.49	0.564
Dissolved oxygen (DO), mg/l	264	1.2-17.0	10.6	10.9
pH	310	5.0-9.7	7.3	7.37
Total phosphorus (P), mg/l	247	0-16.0	0.031	0.130
PO ₄ -P, mg/l	256	0-0.49	0.020	0.0373
Specific conductance, umhos/cm	217	120-295	190.0	188.0
Fecal streptococci, colonies/100 ml	197	16.0-38,000	300.0	1080.0
Total streptococci, colonies/100 ml	104	7.0-70,000	790.0	2750.0
Total dissolved solids (TDS), mg/l	4	40-171.0	95.0	100.0
Biochemical oxygen demand (BOD5), mg/l	51	0-4.6	1.0	1.08

Table 2

Summary of Water Quality Data Taken at Schnecksville Gage (Dam site)

Variable	Sample Size	Range	Median	Mean
Alkalinity, mg/l	38	14-80	31.0	37.3
Total organic carbon (TOC), mg/l	30	0-34	2.9	4.04
Total coliforms, colonies/100 ml	45	6.0-14,000	1200.0	2290.0
Fecal coliforms, colonies/100 ml	48	3.0-9,600	280.0	654.0
NO ₃ -N, mg/l	51	1.1-5.4	3.39	3.26
NO ₂ -N, mg/l	7	0.005-0.03	0.01	0.0143
NH ₄ -N, mg/l	49	0-1.0	0.05	0.101
Total nitrogen (N), mg/l	5	2.7-4.41	3.05	3.25
Total Kjeldahl nitrogen (TKN), mg/l	48	0.04-1.9	0.505	0.583
Organic nitrogen (N), mg/l	41	0.04-1.8	0.41	0.519
Dissolved oxygen (DO), mg/l	49	5.0-17.0	11.4	11.4
pH	56	6.3-9.2	7.4	7.48
Total phosphorus (P), mg/l	47	0.008-0.39	0.03	0.0484
PO ₄ -P, mg/l	49	0-0.086	0.01	0.0181
Specific conductance, umhos/cm	43	140-260	185.0	189.0
Fecal streptococci, colonies/100 ml	22	16-20,000	219.0	1440.0
Total streptococci, colonies/100 ml	21	7-11,000	540.0	1710.0
Biochemical oxygen demand (BOD ₅), mg/l	6	0-3.0	1.0	1.18

Table 3
Summary of Water-Quality Data for Reservoir Tributaries

Variable	Switzer Creek		Jordan Creek (Upstream)		Mill Creek		Jordan Creek (Dam site)		Lyon Creek	
	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
Alkalinity, mg/l	32.2	13.7	<u>49.4</u>	79.2	39.6	19.3	37.3	18.2	38.8	17.2
Total organic carbon (TOC), mg/l	3.3	1.9	3.4	2.1	<u>4.6</u>	5.0	4.0	5.6	4.0	3.3
Total coliforms, colonies/100 ml	3517.0	4435.0	2546.0	2838.0	<u>13,930.0</u>	43,921.0	2280.0	3003.0	8374.0	11,166.0
Fecal coliforms, colonies/100 ml	1285.0	3655.0	708.0	1791.0	1,033.0	1,390.0	653.0	1451.0	<u>4044.0</u>	7,169.0
Flow, m ³ /sec	0.53	0.59	0.89	1.28	0.32	0.37	<u>3.26</u>	3.74	1.73	9.0
NO ₃ -N, mg/l	3.34	1.44	2.99	1.47	<u>4.4</u>	1.84	3.25	1.16	3.92	1.5
NO ₂ -N, mg/l	<u>0.028</u>	0.0076	0.01	0.009	0.013	0.007	0.014	0.0084	0.012	0.0078
NH ₄ -N, mg/l	0.21	0.74	0.14	0.42	0.1	0.17	0.1	0.17	<u>0.23</u>	0.82
Total nitrogen (N), mg/l	3.0	--	2.87	0.79	<u>4.2</u>	1.15	3.25	0.71	3.05	1.24
Total Kjeldahl nitrogen (TKN), mg/l	0.71	0.75	0.65	0.50	0.62	0.44	0.58	0.37	<u>0.817</u>	0.91
Total phosphorus (P), mg/l	0.045	0.05	0.033	0.032	<u>0.49</u>	2.34	0.048	0.064	0.063	0.08
PO ₄ -P, mg/l	0.018	0.025	0.016	0.014	<u>0.11</u>	0.0095	0.018	0.017	0.029	0.038
Specific conductance, umhos/cm	184.0	19.1	154.0	23.0	<u>215.0</u>	29.8	189.0	23.6	199.0	28.6
Fecal streptococci, colonies/100 ml	899.0	1569.0	793.0	1340.0	1,078.0	2,464.0	1437.0	4416.0	<u>1821.0</u>	5,753.0
Total streptococci, colonies/100 ml	1333.0	1781.0	1374.0	1842.0	2,199.0	4,515.0	1708.0	2778.0	<u>7353.0</u>	18,347.0

Note: Maximum values for each constituent are underlined.

Table 4
Morphometric Characteristics for Proposed Trexler Lake
and Three Other CE Impoundments

<u>Morphometric Characteristic</u>	<u>Trexler</u>	<u>Beltzville</u>	<u>Prompton</u>	<u>Francis E. Walter</u>
Water surface elevation, m msl	150.4	191.4	343.1	396.2
Volume, m ³	5.06×10^7	5.09×10^7	4.19×10^6	2.47×10^6
Surface area, ha	493.0	383.0	113.0	36.4
Maximum depth, m	31.4	34.8	11.90	18.3
Mean depth, m	10.3	13.3	3.70	6.8
Mean annual flow, m ³ /sec	2.61	2.28	0.57	--
Residence time, yr	0.62	0.71	0.23	--
Length, km	13.9	11.3	4.0	2.7
Area of drainage basin, km ²	135.0	242.0	155.4	747.0

Table 5
Surrounding Impoundments Considered in the Study

<u>Lake</u>	<u>County</u>	<u>State</u>
Beltzville (CE)	Carbon	Pa.
Conewago	York	Pa.
Francis E. Walter (CE)	Carbon, Montgomery	Pa.
Green Lane	Montgomery	Pa.
Hopewell	Berks	Pa.
Marburg	York	Pa.
Marsh Creek	Chester	Pa.
Nockamixon	Bucks	Pa.
Prompton (CE)	Wayne	Pa.
Round Valley	Hunterdon	N. J.
Shawnee	Bedford	Pa.
Spruce Run	Hunterdon	N. J.

Table 6
Phytoplankton Composition in Surrounding Impoundments

<u>Cyanophyta</u> <u>(Blue-Green Algae)</u>	<u>Chlorophyta</u> <u>(Green Algae)</u>	<u>Chrysophyta</u> <u>(Diatoms)</u>	<u>Pyrrophyta</u> <u>(Dino-Flagellates)</u>
<i>Aphanizomenon</i> sp.	<i>Chlamydomonas</i> sp.	<i>Asterionella</i> sp.	<i>Ceratiwm</i> sp.
<i>Anabaena</i> sp.	<i>Closterium</i> sp.	<i>Dinobryon</i> sp.	
<i>Anacystis</i> sp.	<i>Dictyosphaerium</i> sp.	<i>Fragilaria</i> sp.	
<i>Aphanocapsa</i> sp.	<i>Gloeocystis</i> sp.	<i>Melosira</i> sp.	
<i>Gomphosphaeria</i> sp.	<i>Pediastrum</i> sp.	<i>Synedra</i> sp.	
<i>Microcystis</i> sp.	<i>Stauroastrum</i> sp.		
<i>Lyngbya</i> sp.	<i>Sphaerocystis</i> sp.		
<i>Phormidium</i> sp.			
<i>Synechocystis</i> sp.			

Table 7
Chlorophyll a Concentrations Measured in Surrounding Lakes

<u>Lake</u>	<u>Date of Sampling</u>	<u>Chlorophyll a</u> <u>Concentration</u> <u>mg/m³</u>
Beltzville	17 Apr 73	3.5-5.1
	24 Jul 73	5.7-8.4
	4 Oct 73	2.8-3.5
Shawnee	14 Jul 71	13.0
Marsh Creek*	10 Jul 74	4.3
Hopewell	11 Jul 73	1.9
Nockamixon*	30 Jul 74	4.1
Conewago	9 Jul 74	3.0

* 1 year old at the time of measurement.

Table 8
Summary of the Major Modifications Made
by WES to WQRRS Reservoir Model

<u>Modification</u>	<u>Description</u>
1	The BOD compartment was replaced by a dissolved organic (DOR) compartment because BOD values may include effects of nitrification, decay of particulate organic matter, and algal respiration, which are included in other parts of the reservoir model.
2	The diffusion coefficient was modified to be a function of wind speed so that effects of wind mixing could be simulated.
3	The basis for determining reservoir stratification stability was changed from temperature to density differences so that inverse stratifications at temperatures below 4°C could be simulated.
4	The calculation of the surface layer volume was modified to be of variable volume, instead of assuming it to be of constant volume, to ensure accurate mass balances.
5	The inflow algorithm was modified so that under isothermal conditions, the inflow could be placed on the surface or at the bottom when the inflow differed in density from the isothermal reservoir by more than a specified amount.
6	A predator was added to the fish compartment, and the planktivore was modified to feed on detritus also.
7	The calculation of production was corrected to correspond to net primary production above the 1 percent light level.
8	The fraction of solar radiation absorbed in the surface layer was made variable.
9	The zooplankton compartment was modified so that they feed on both detritus and algae.
10	The model was modified so it could be run in a Monte Carlo mode, with specified distributions for coefficients and updates.

Table 9
Mean Monthly Flows in Jordan Creek
at Allentown, Pennsylvania

Month	1969		1973		1974		30-Year Mean 1945-1974	
	m ³ /sec	cfs	m ³ /sec	cfs	m ³ /sec	cfs	m ³ /sec	cfs
Jan	1.8	63	5.6	198	5.3	186	3.1	109
Feb	1.8	62	6.4	225	3.4	119	4.2	149
Mar	2.2	79	3.5	125	5.5	193	5.2	184
Apr	3.7	131	7.8	277	8.1	285	4.0	140
May	1.3	47	4.6	163	2.2	78	2.6	91
Jun	1.1	39	4.6	164	1.0	34	1.4	48
Jul	1.7	61	2.5	88	1.2	43	0.9	30
Aug	3.1	111	0.9	33	1.2	43	0.9	31
Sep	1.1	40	1.1	38	2.6	90	0.9	32
Oct	1.0	34	0.7	26	1.9	68	0.8	28
Nov	1.2	42	0.9	31	1.3	44	2.0	71
Dec	2.8	97	9.5	336	5.5	194	3.2	113
Mean Annual	1.9	67	4.0	142	3.3	115	2.9	104

Table 10
General Regulation Scheme for the Selective
Withdrawal Structure Used in Study

<u>Port</u>	<u>Period of Operation, Julian days</u>
1	60-140
2	135-190
3	150-230
4	200-290
5	220-310
6	290-60
Floodgates	When needed

Table 11
Summary of Sensitivity Simulations

Run No.	Variable (Mnemonic Name)	Modification
74-1	Light half-saturation coefficient (PS2L)	Decreased PS2L(1) from 4 kcal/m ² /hr to 2 and PS2L(2) from 8 kcal/m ² /hr to 4.
74-2	Light half-saturation coefficient	Increased PS2L(1) from 4 kcal/m ² /hr to 8 and PS2L(2) from 8 kcal/m ² /hr to 12.
74-3	Phosphorus half-saturation coefficient (PS2PO4)	Increased PS2PO4(1) from 0.006 mg/l to 0.014 and PS2PO4(2) from 0.003 mg/l to 0.006.
74-4	Carbon half-saturation coefficient (PS2C02)	Decreased PS2C02(1) and PS2C02(2) from 0.10 mg/l to 0.05.
73-1	Carbon half-saturation coefficient	Decreased PS2C02(1) and PS2C02(2) from 0.10 mg/l to 0.05.
73-2	Algae settling rate (TSETL)	Increased TSETL(1) from 0.1 m/day to 0.15 and TSETL(2) from 0.3 m/day to 0.4.
73-3	Algae settling rate	Decreased TSETL(1) from 0.1 m/day to 0.05 and TSETL(2) from 0.3 m/day to 0.2.
74-5	Phosphorus inflow (PO4IN)	Doubled PO4IN
73-4	Phosphorus inflow	Doubled PO4IN
69-1	Phosphorus inflow	Doubled PO4IN
74-6	Phosphorus inflow	Halved PO4IN
73-5	Phosphorus inflow	Halved PO4IN
69-2	Phosphorus inflow	Halved PO4IN
73-6	Phosphorus inflow	PO4IN = 0.018 mg/l

(Continued)

Table 11 (Concluded)

Run No.	Variable (Mnemonic Name)	Modification
74-7	Coliform inflow (COLIN)	COLIN = COLIN \times 10.
73-7	Coliform inflow	COLIN = COLIN \times 10.
69-3	Coliform inflow	COLIN = COLIN \times 10.

Table 12

Summary of Coefficients Varied in Monte Carlo Simulation

Variable*	Distribution	Minimum	Maximum	Mean	Standard Deviation
PS2C02(1)	Uniform	0.05	0.15	--	--
PS2C02(2)	Uniform	0.05	0.15	--	--
PS2L(1)	Uniform	1.0	4.0	--	--
PS2L(2)	Uniform	2.0	6.0	--	--
PS2N(1)	Rotated Log Normal	0.001	0.046	0.018	0.014
PS2N(2)	Rotated Log Normal	0.001	0.044	0.016	0.014
PS2P04(1)	Rotated Log Normal	0.002	0.010	0.006	0.002
PS2P04(2)	Rotated Log Normal	0.001	0.0051	0.003	0.001
TNH3DK	Uniform	0.12	0.18	--	--
TN02DK	Uniform	0.35	0.41	--	--
TCOLDK	Uniform	1.20	1.50	--	--
TDFTDK	Uniform	0.015	0.023	--	--
TDORDK	Uniform	0.1	0.16	--	--
TDSETL	Uniform	0.05	0.3	--	--
TPMAX(1)	Normal	1.2	3.6	2.44	1.21
TPMAX(2)	Normal	1.7	2.5	2.10	0.37
TPRESP	Normal	0.02	0.32	0.17	0.075
TSETL(1)	Uniform	0.0	0.2	--	--
TSETL(2)	Uniform	0.08	0.5	--	--
TZMAX	Normal	0.48	0.52	0.505	0.01
TZMORT	Normal	0.0031	0.007	0.005	0.001
TZRESP	Normal	0.12	0.28	0.2	0.04

* See Table C2 for description of variable.

Table 13
Summary of Update Randomization Data
Varied in Monte Carlo Simulation

<u>Variable (Mnemonic Name)</u>	<u>Distribution</u>	<u>Minimum</u>	<u>Maximum</u>
Ammonia Inflow (CNH ₄ IN)	Uniform	0.01	0.2
Coliform Inflow (COLIN)	Uniform	20.0	1500.0
Detritus Inflow (DETIN)	Uniform	0.05	10.0
Phosphorus Inflow (PO ₄ IN)	Uniform	0.001	0.04

Table 14
Predicted Average In-Lake Phosphorus Concentrations
and Phosphorus Equations

<u>Source</u>	<u>Average Annual In-Lake P Concentration, mg/m³</u>			<u>Equation</u>
	<u>1969</u>	<u>1973</u>	<u>1974</u>	
Dillon and Rigler (1974)	32	60	36	$\{P\} = \frac{L(1 - R)}{z\rho}$
Larsen and Mercier (1976)	35	67	40	$\{P\} = \{\bar{p}\}(1 - R_p)$
Vollenweider (1976)	35	67	39	$\{P\} = \frac{L}{q_s \left\{ 1 + \left(\frac{\bar{z}}{q_s} \right) \right\}}$

NOTE: $\{P\}$ = average phosphorus concentration, mg/m³
 L = annual phosphorus loading, mg/(m² · yr)
 R, R_p = retention coefficients
 \bar{z} = mean depth, m
 ρ = flushing rate per year, yr⁻¹
 $\{\bar{p}\}$ = average influent P concentration
 q_s = areal water load or Q/A
 Q = annual outflow, m³/yr
 A = surface area of impoundments, m²

Table 15
Retention Coefficients or Estimates of Sedimentation
and the Retention Equations

Source	Retention Coefficients			Equation
	1969	1973	1974	
Kirchner and Dillon (1975)	0.54	0.48	0.50	$R = 0.426 \exp(-0.271 q_s) + 0.574 \exp(-0.00949 q_s)$
Larsen and Mercier (1976)	0.50	0.42	0.45	$R_p = \frac{1}{1 + \rho^{1/2}}$

Table 16
Predicted Chlorophyll Concentrations for Proposed Trexler Lake
Based on Estimated Phosphorus Concentrations

Source	Chlorophyll <u>a</u> Concentration mg/m ³			Chlorophyll Equation
	1969	1973	1974	
Dillon and Rigler (1974)	11.0	27.4	13.1	$\log_{10} \{chl\ a\} = 1.45 \log_{10} \{P\} - 1.14$ $\{chl\ a\} = 0.367 \{P\}^{0.91}$
Vollenweider (1976)	9.3	16.8	10.3	
Carlson (1977)	13.2	32.8	15.6	$\ln \{chl\ a\} = 1.449 \ln \{P\} - 2.442$

Note: chl a = chlorophyll a concentration, mg/m³.

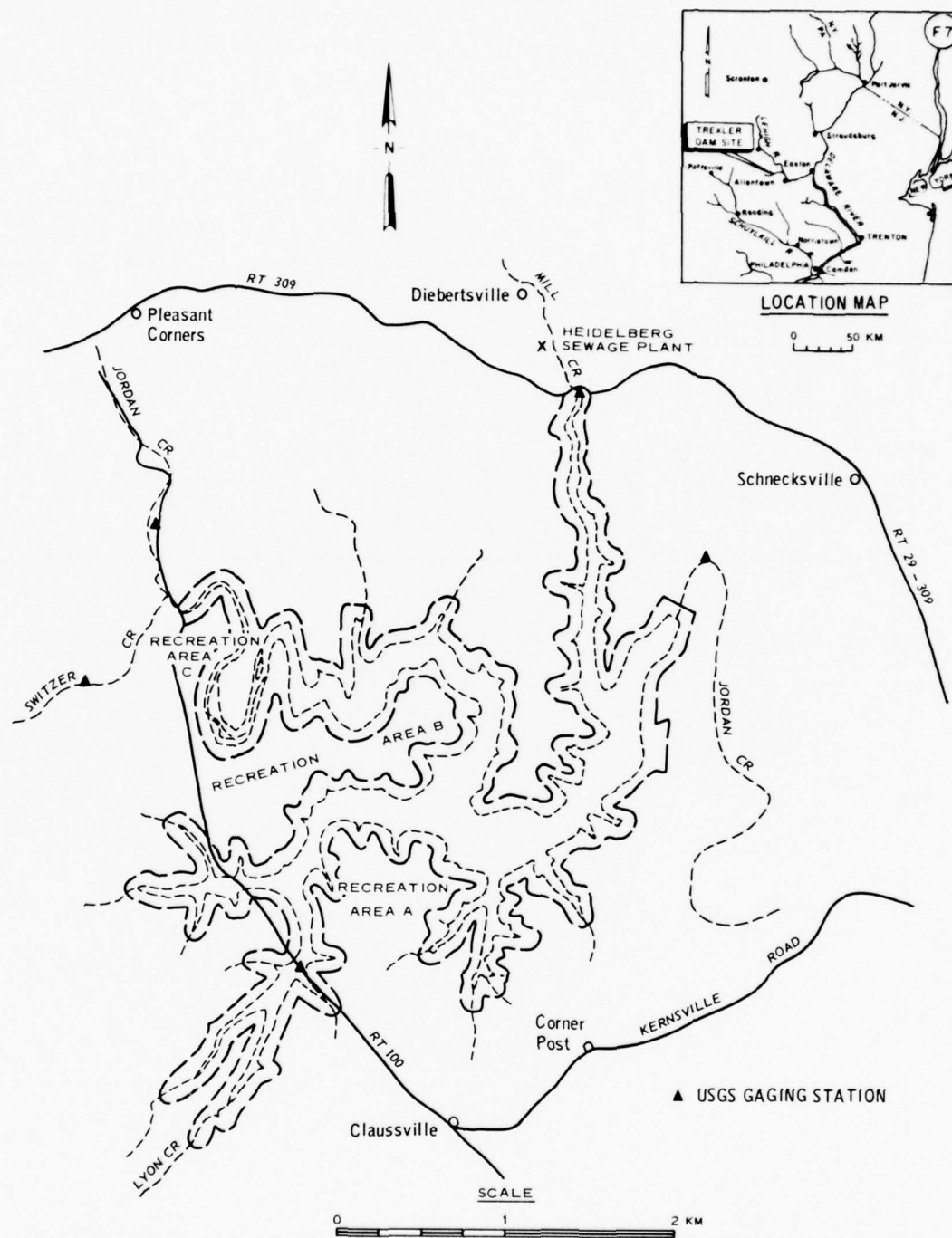


Figure 1. Location and description of project site

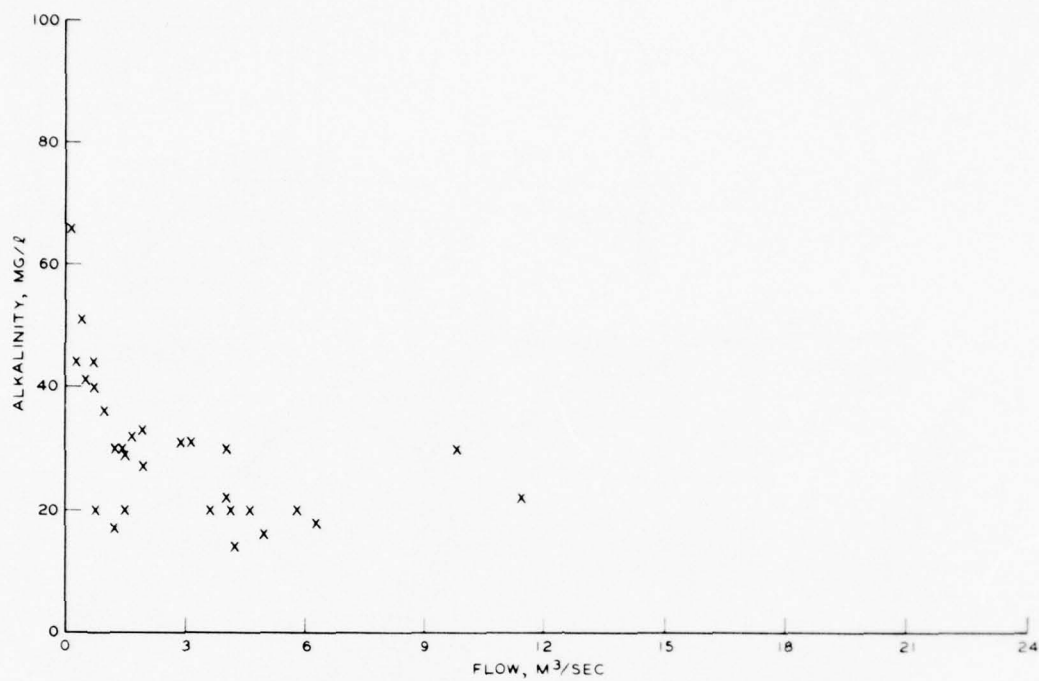


Figure 2. Alkalinity versus flow for all data measured at the damsite

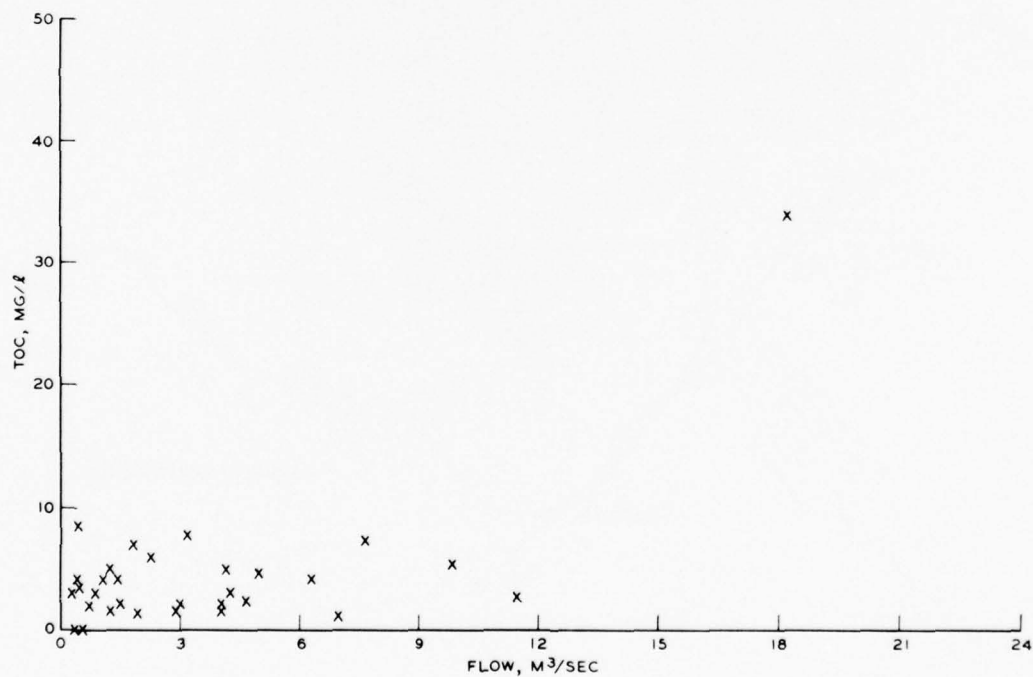


Figure 3. TOC versus flow for all data measured at the damsite

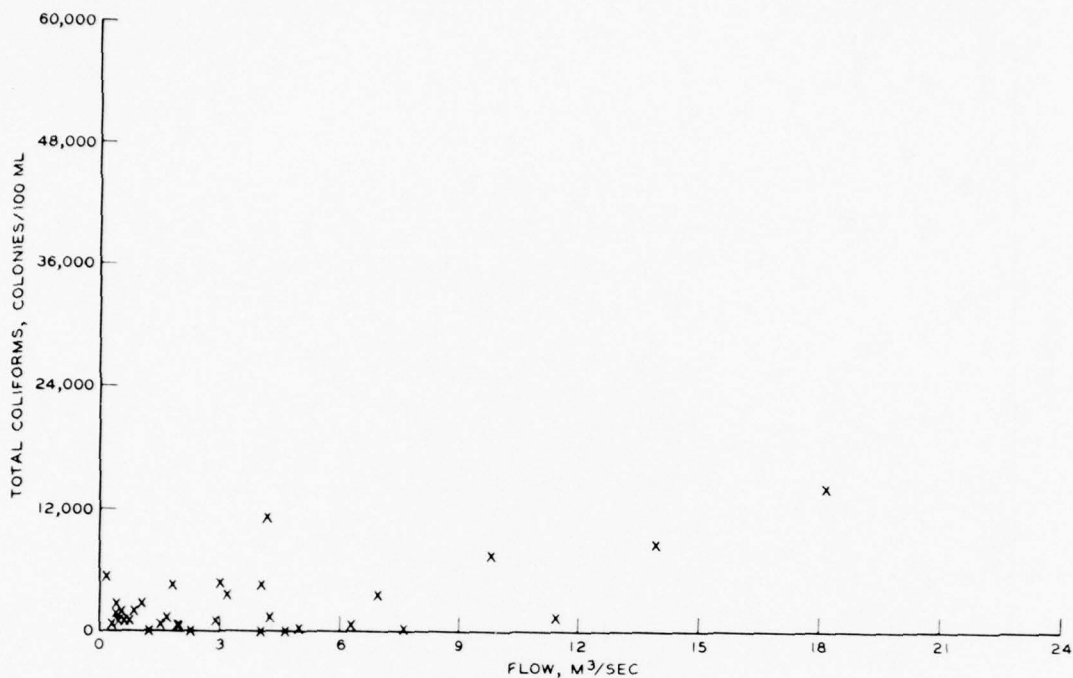


Figure 4. Total coliforms versus flow for all data measured at the damsite

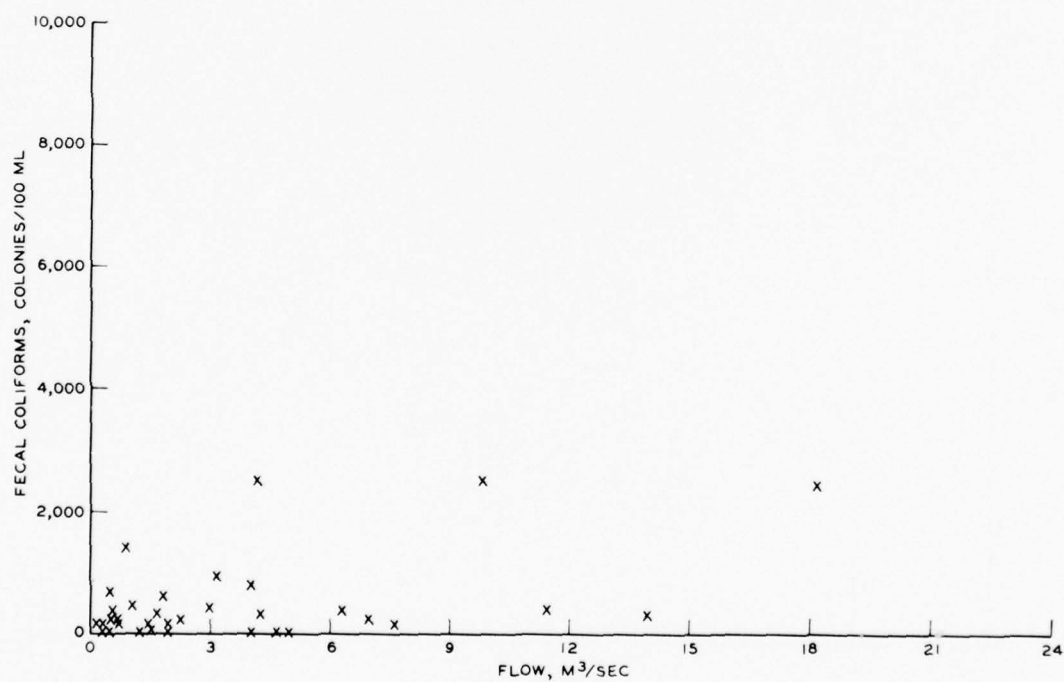
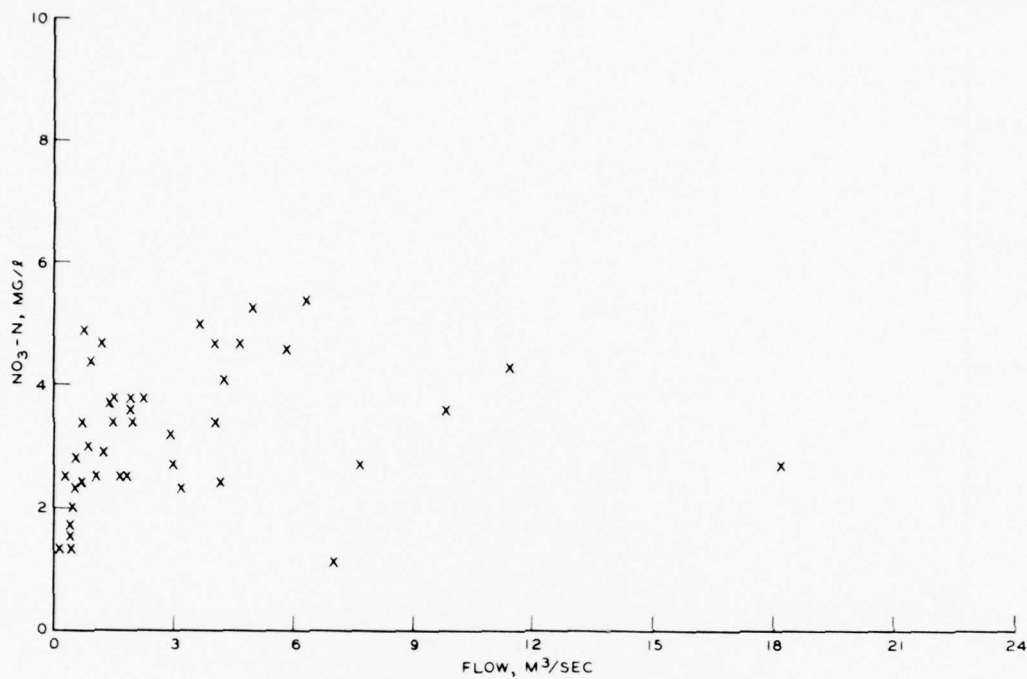


Figure 5. Fecal coliforms versus flow for all data measured at the damsite



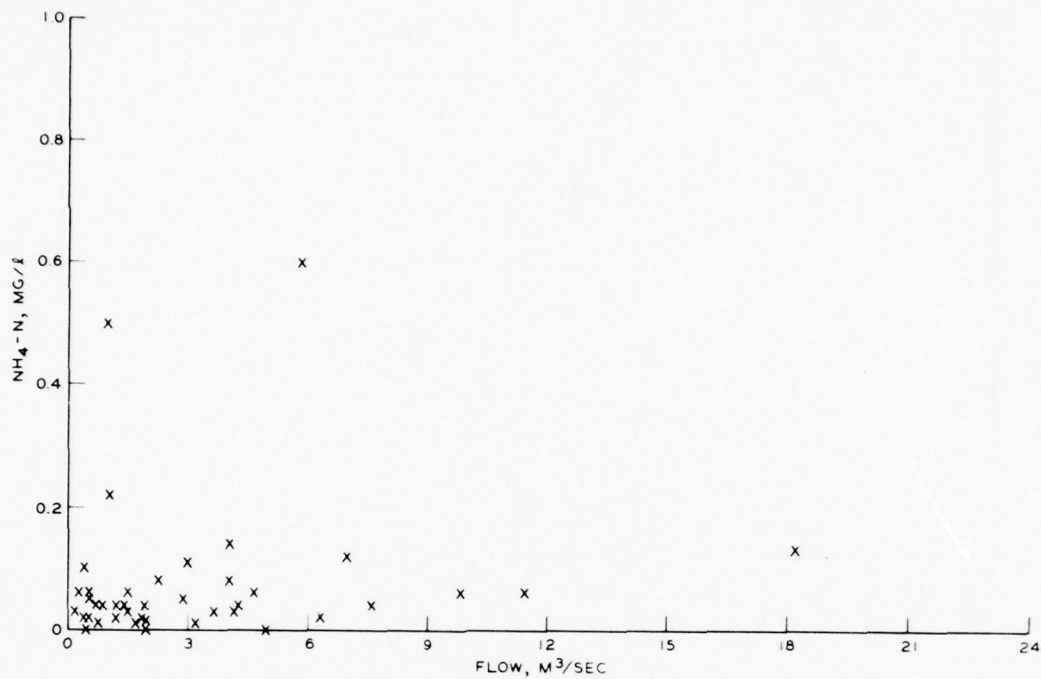


Figure 8. $\text{NH}_4\text{-N}$ versus flow for all data measured at the damsite

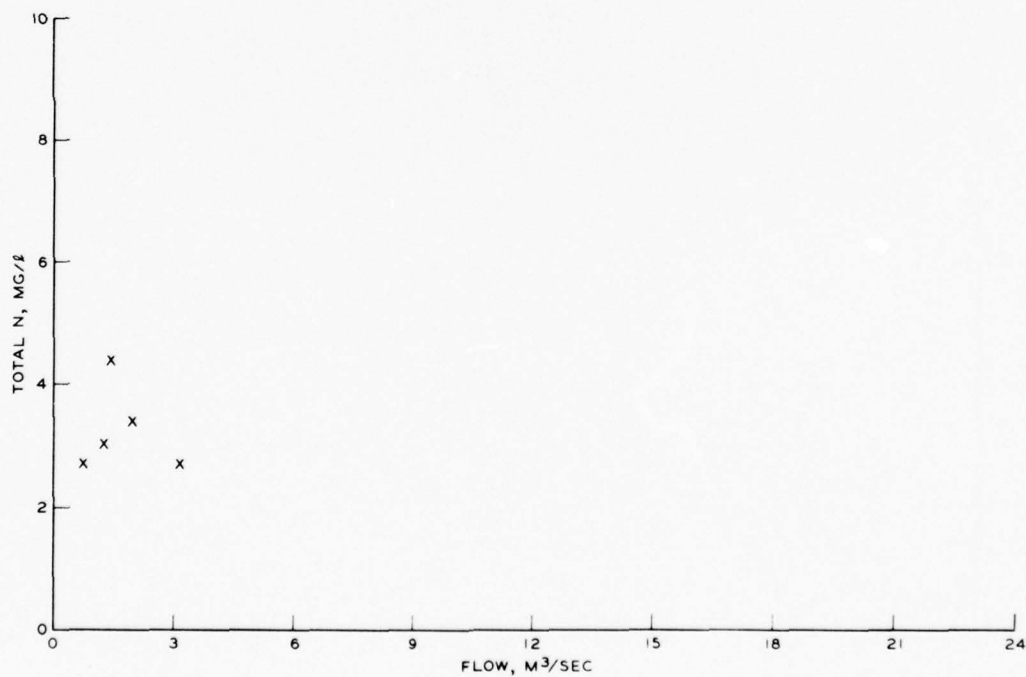


Figure 9. Total N versus flow for all data measured at the damsite

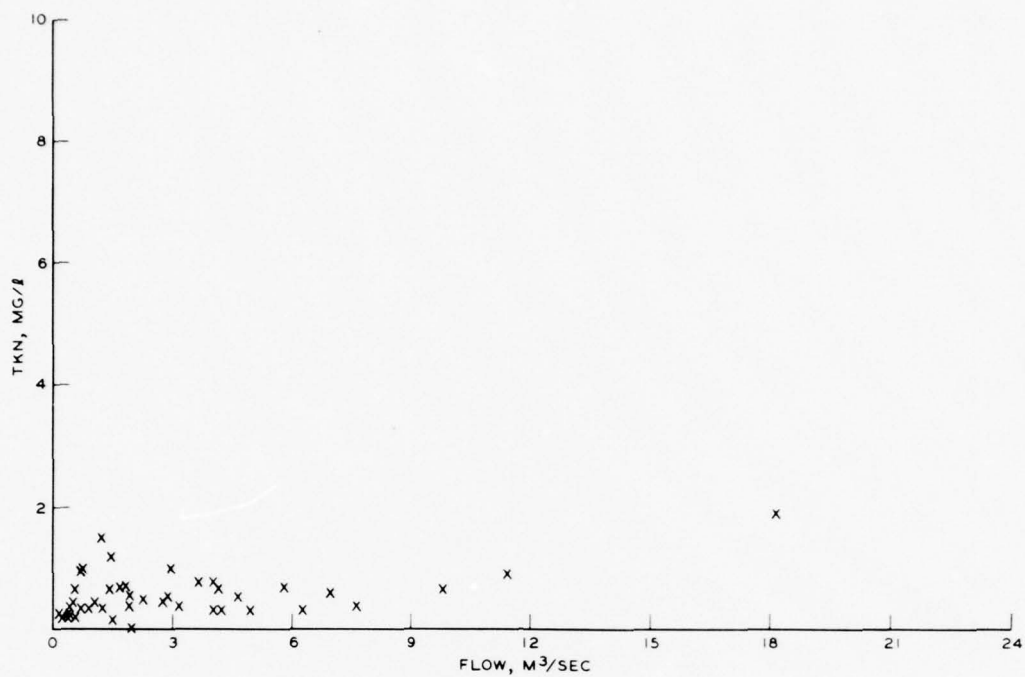


Figure 10. TKN versus flow for all data measured at the damsite

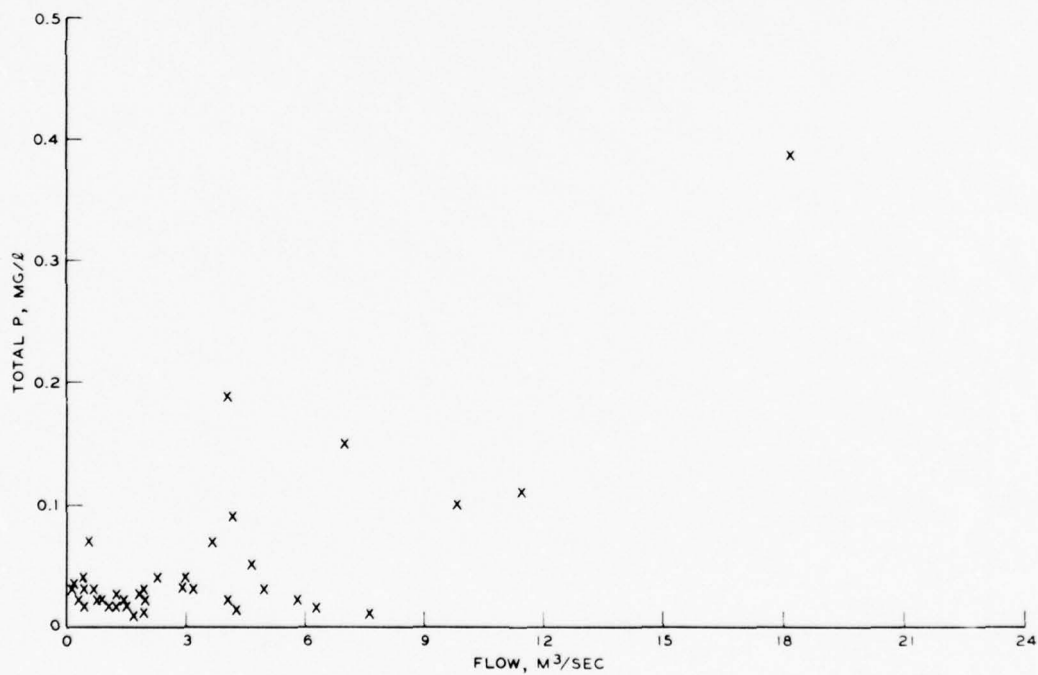


Figure 11. Total P versus flow for all data measured at the damsite

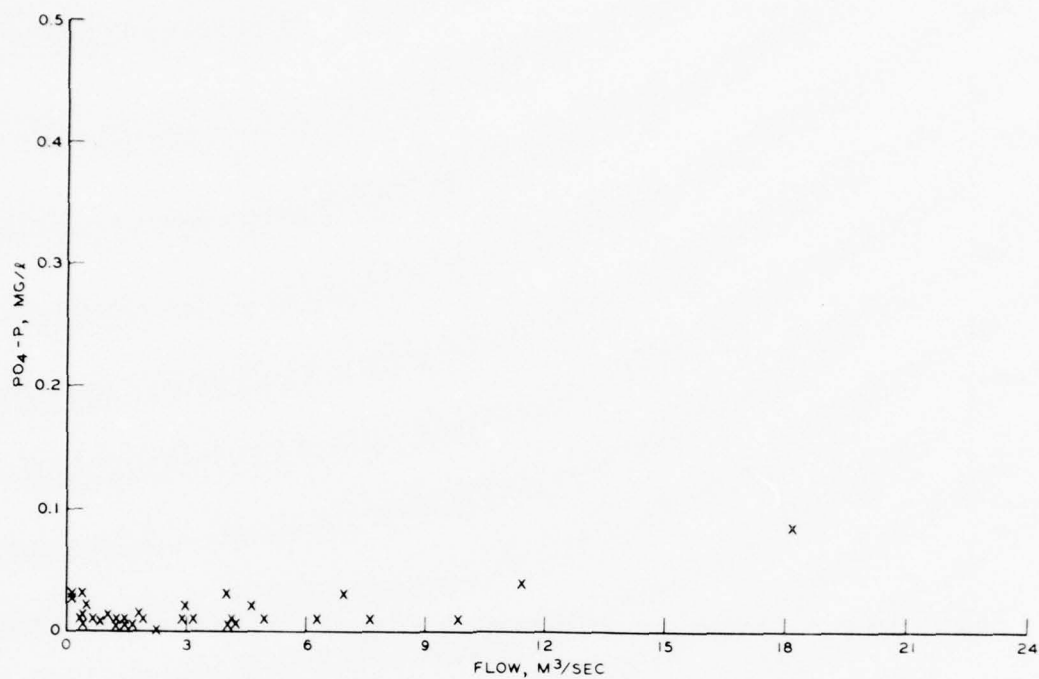


Figure 12. PO₄-P versus flow for all data measured at the damsite

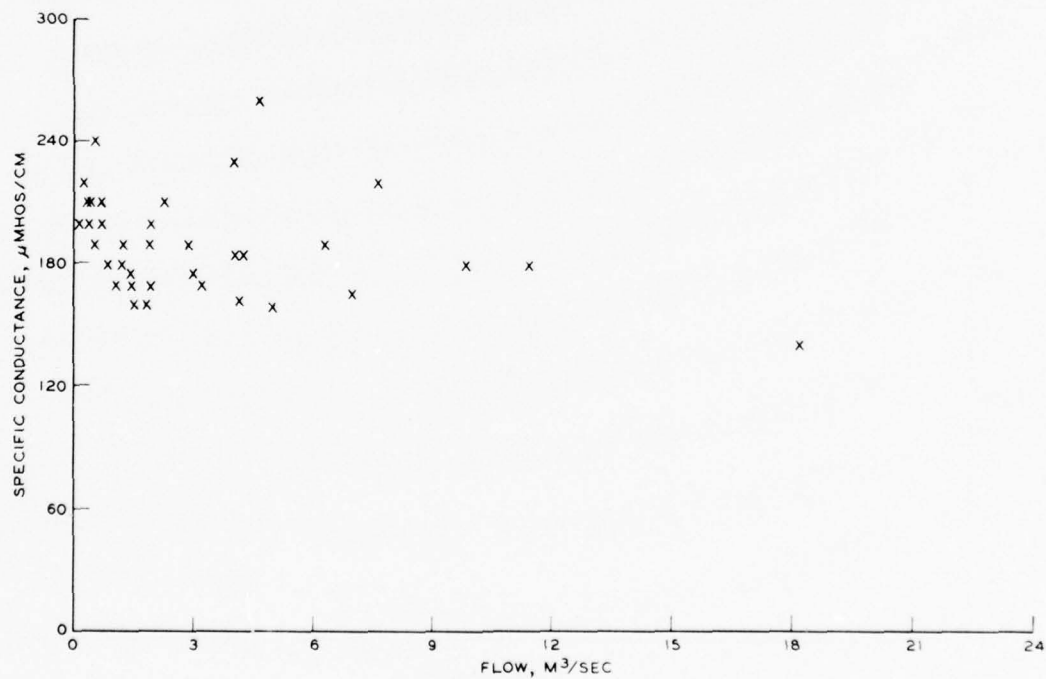


Figure 13. Specific conductance versus flow for all data measured at the damsite

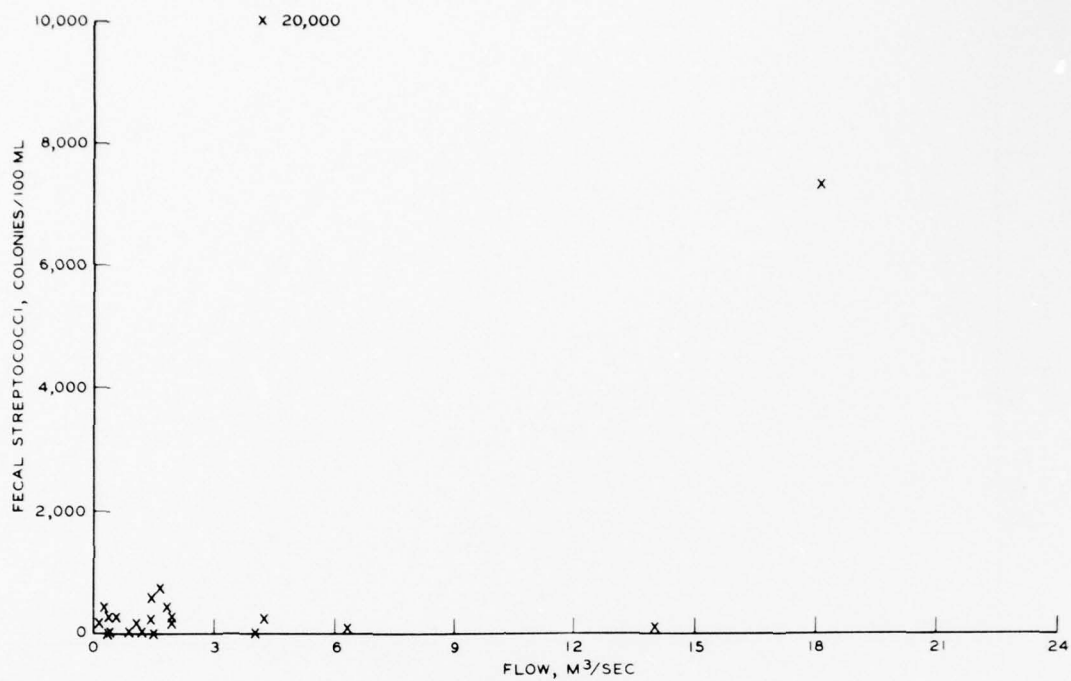


Figure 14. Fecal streptococci versus flow for all data measured at the damsite

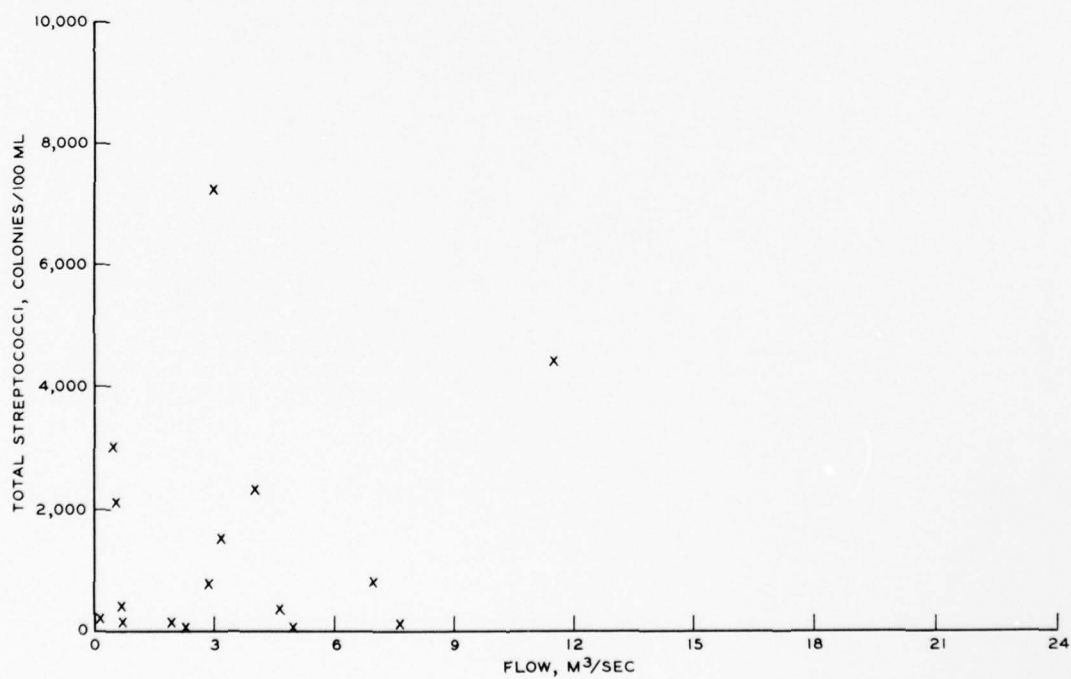


Figure 15. Total streptococci versus flow for all data measured at the damsite

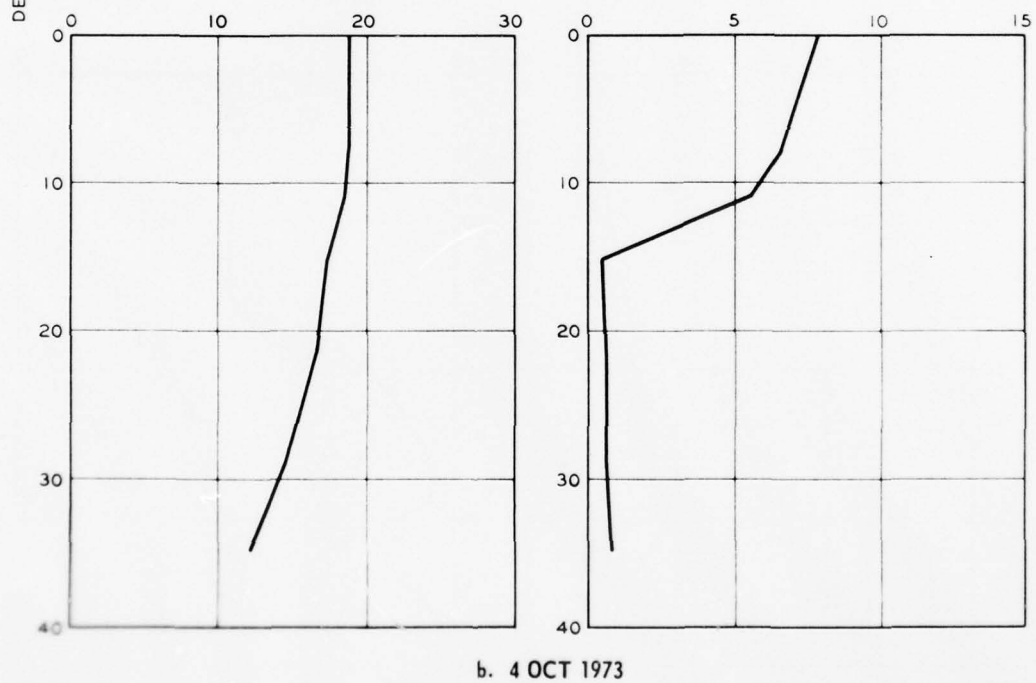
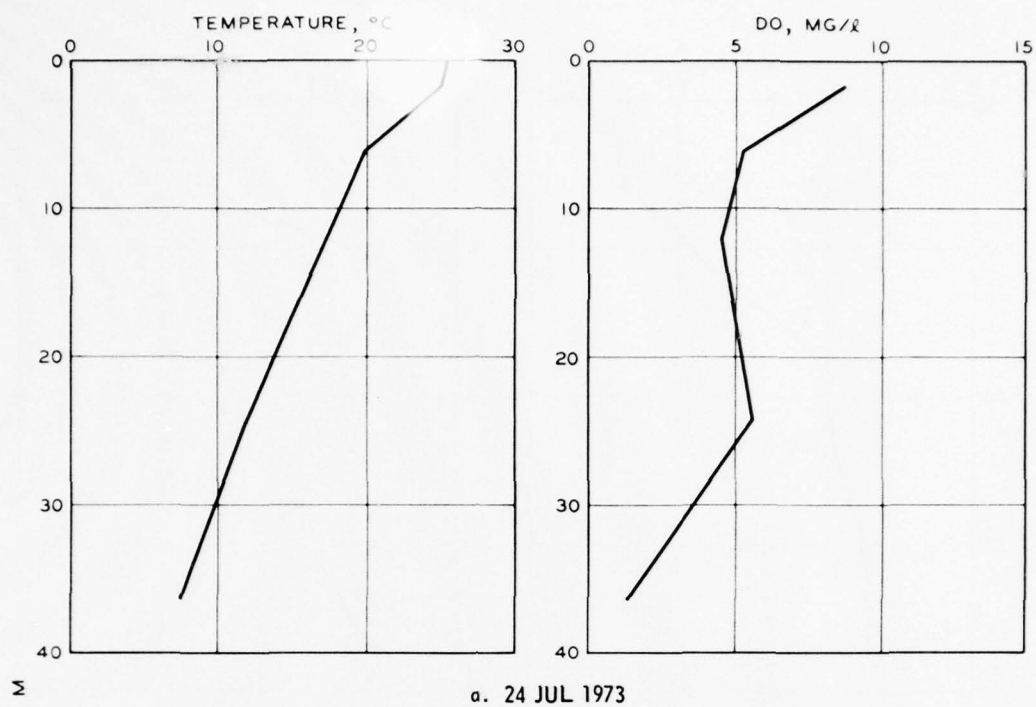
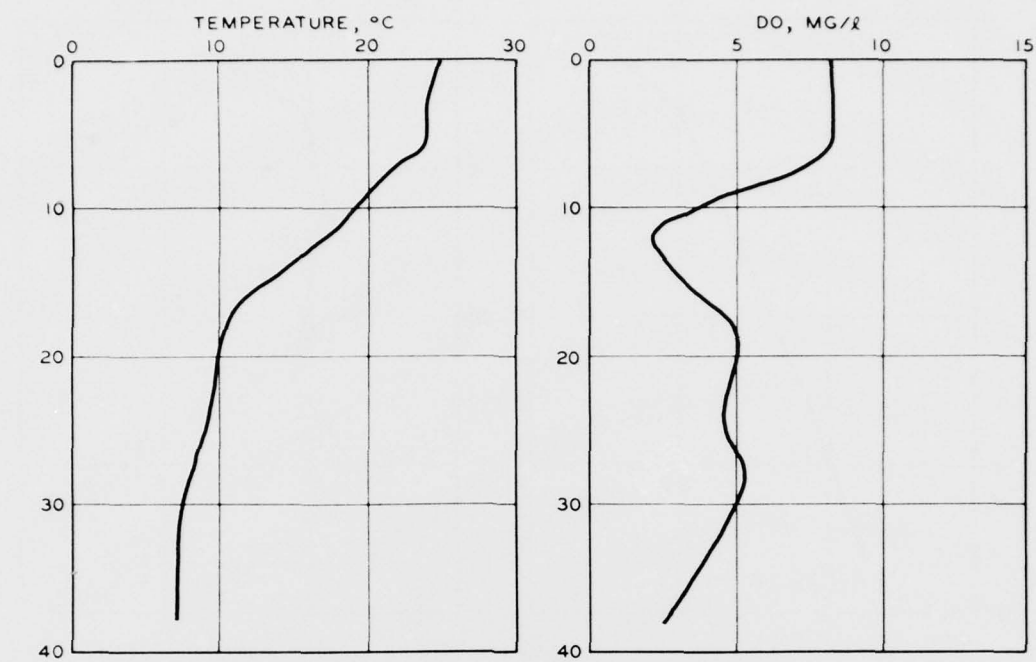
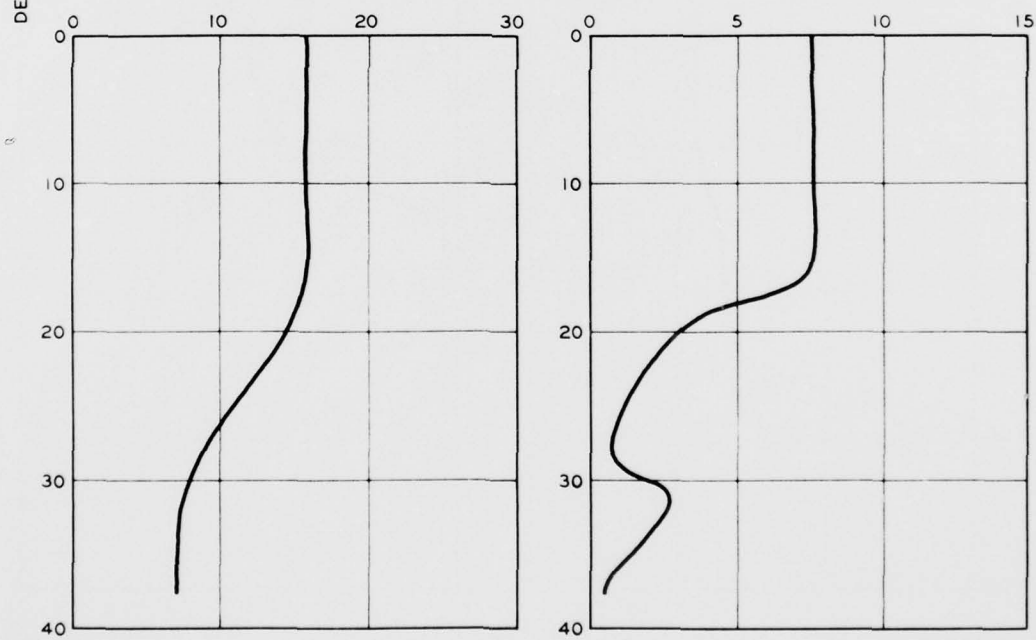


Figure 16. Temperature and DO profiles, Beltzville Lake, 1973



a. 23 AUG 1974 (235)



b. 3 OCT 1974 (276)

Figure 17. Temperature and DO profiles, Beltzville Lake, 1974

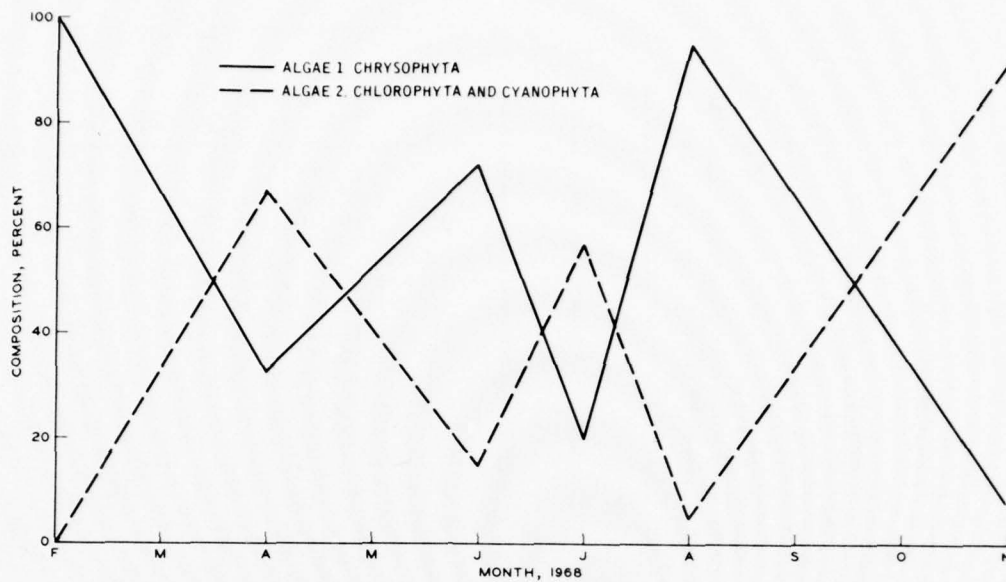


Figure 18. Seasonal composition of algae in Conewago Lake, Pennsylvania

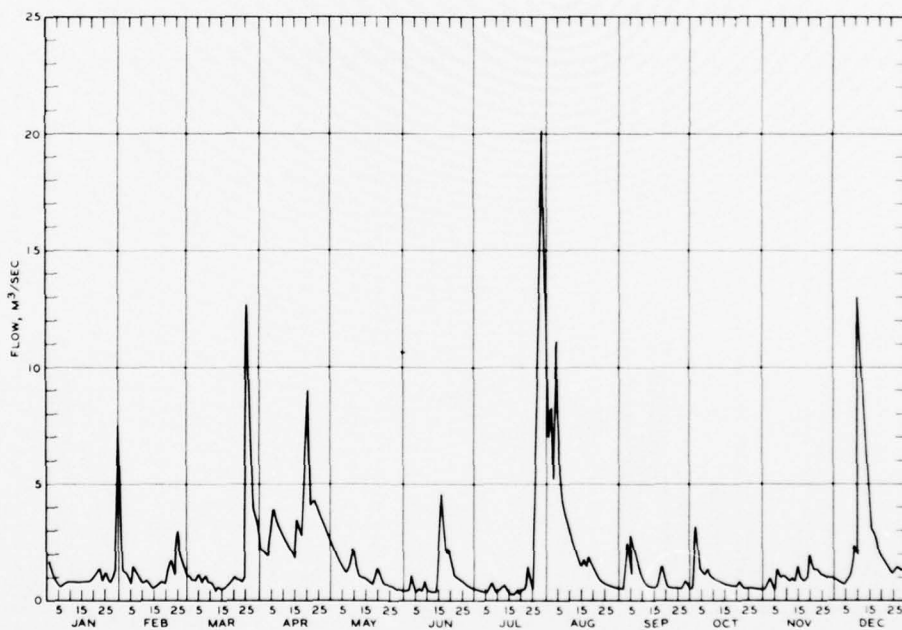


Figure 19. Daily flow at Schnecksville gage for 1969

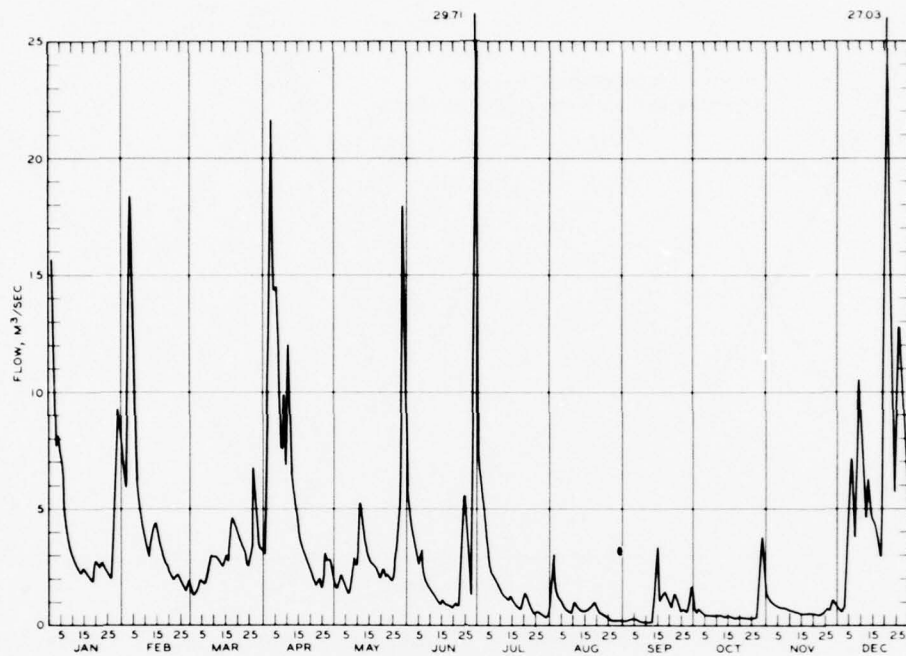


Figure 20. Daily flow at Schnecksville gage for 1973

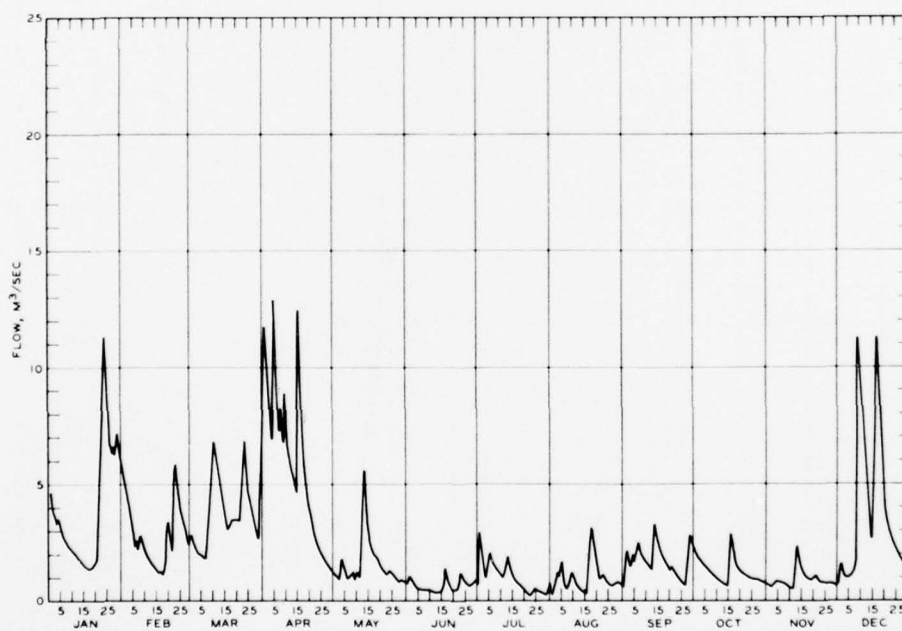


Figure 21. Daily flow at Schnecksville gage for 1974

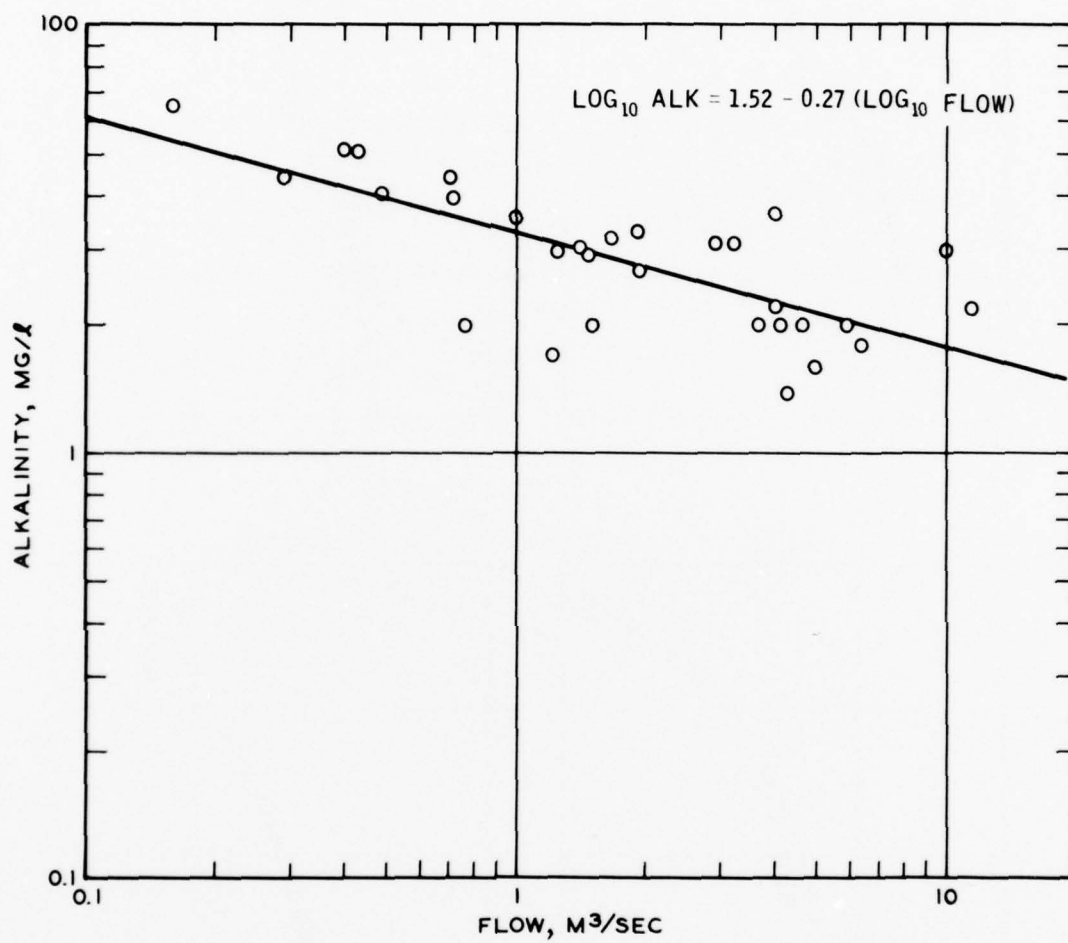


Figure 22. Regression equation and fit of alkalinity versus flow

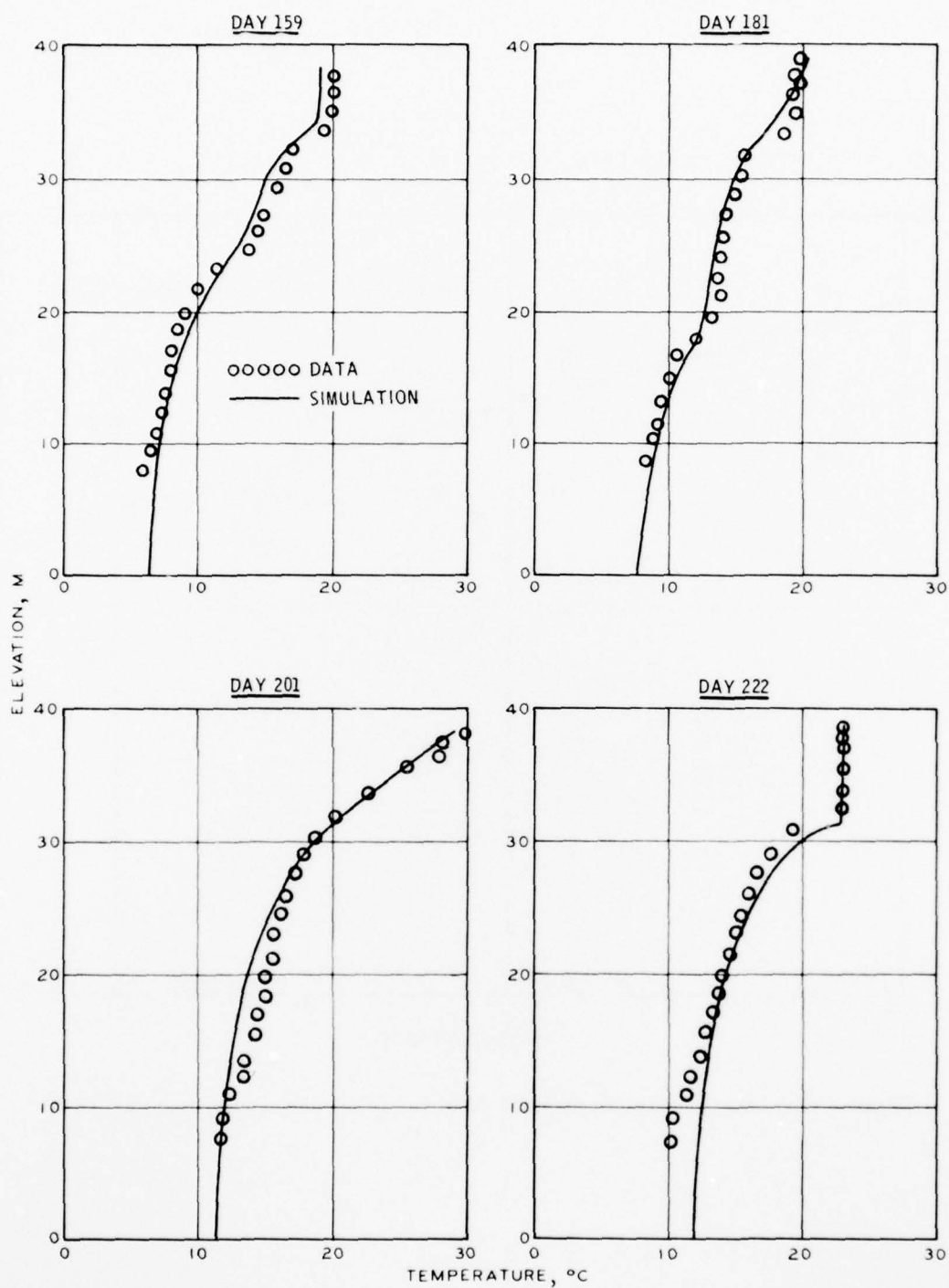


Figure 23. Comparison of actual and simulated temperature profiles at Beltzville Lake, 1972 (sheet 1 of 2)

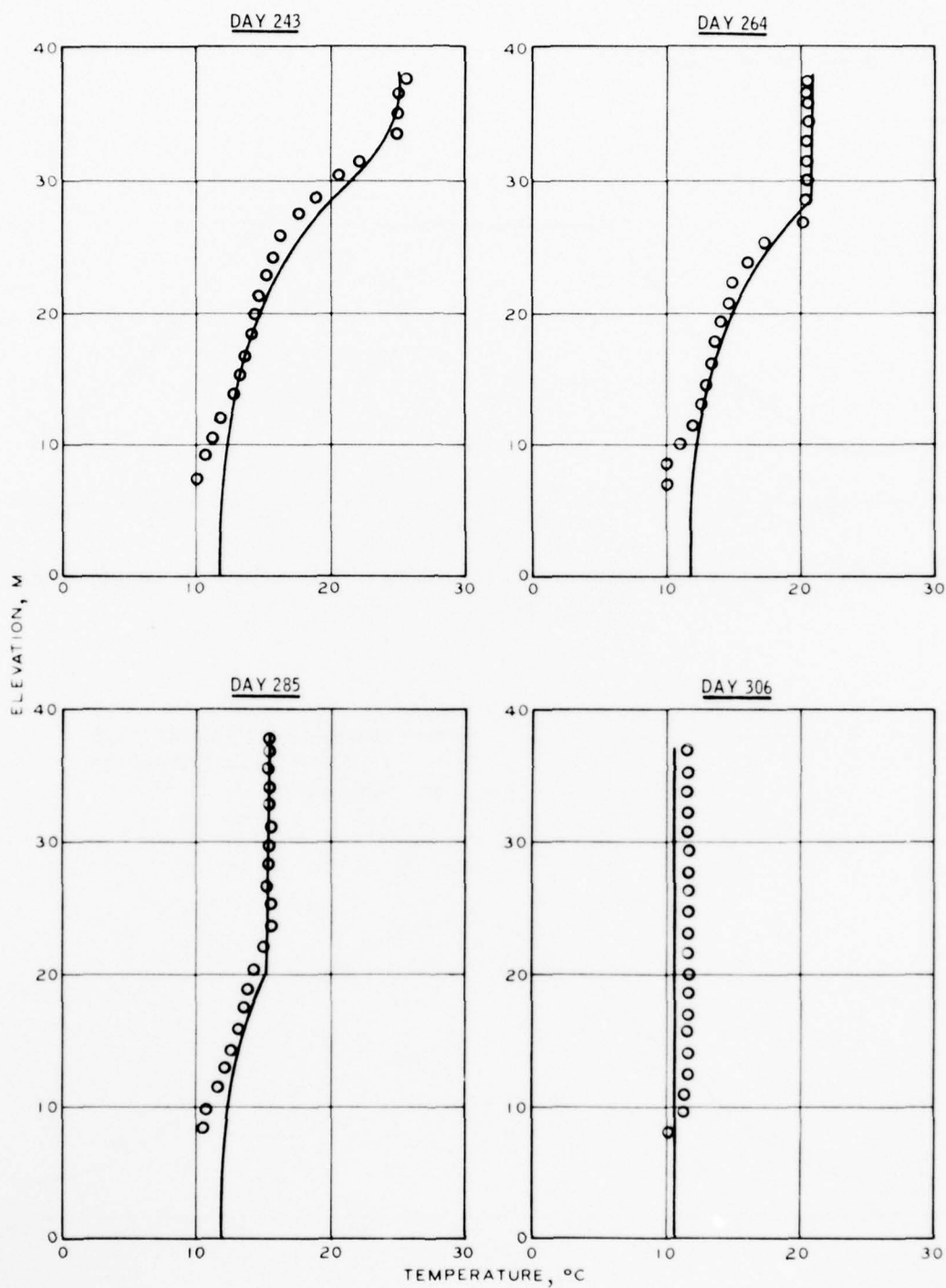


Figure 23 (sheet 2 of 2)

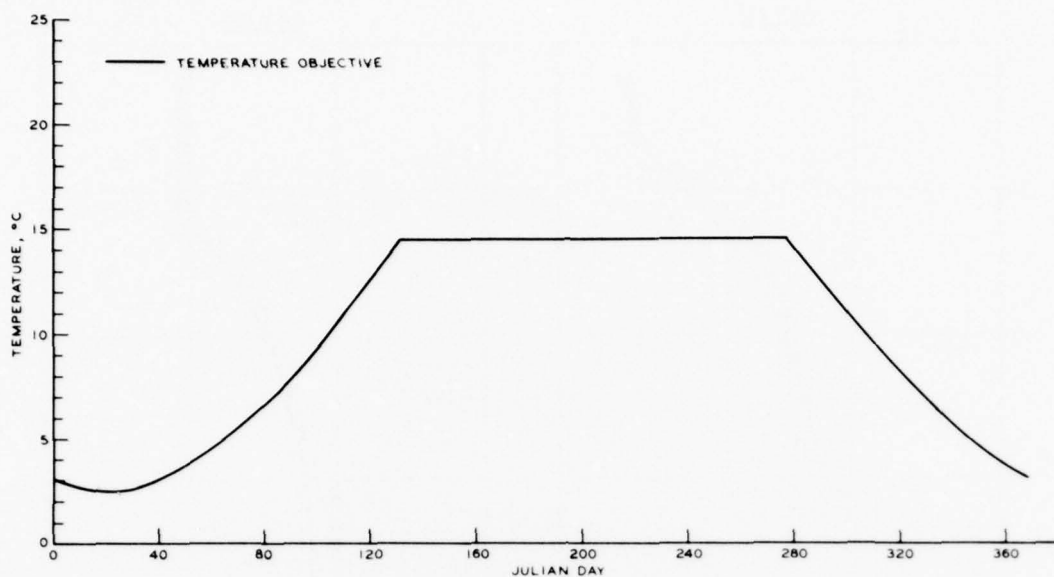


Figure 24. Temperature objective for Jordan Creek based on Pennsylvania water quality criteria

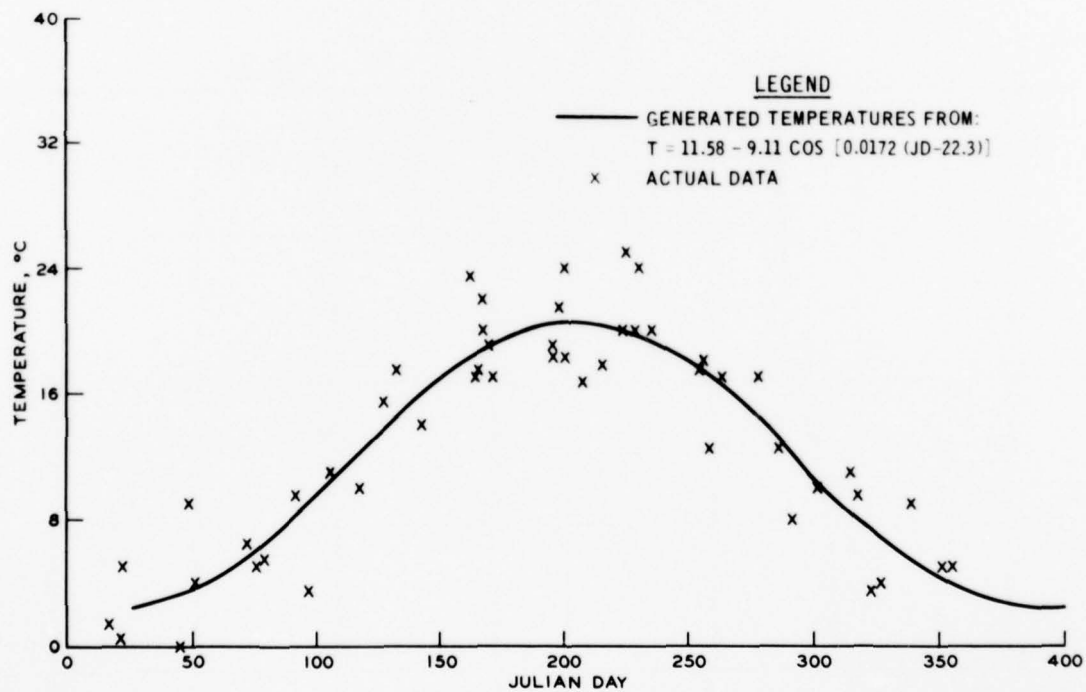


Figure 25. Generated versus actual temperature data at the Schnecksville gage or the dams site

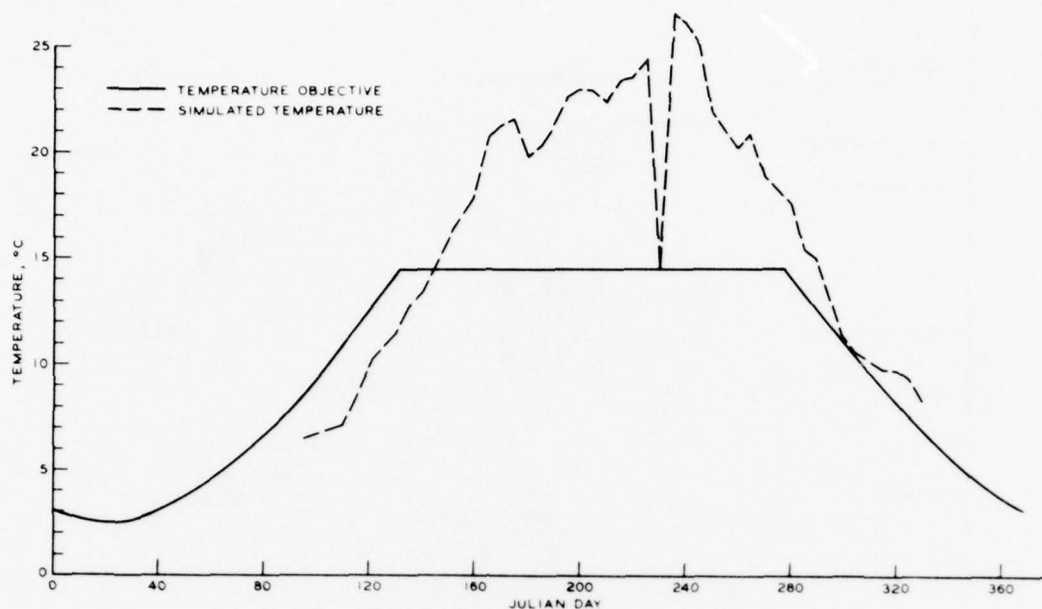


Figure 26. Comparison of temperature objective with simulated release temperature using three ports without blending

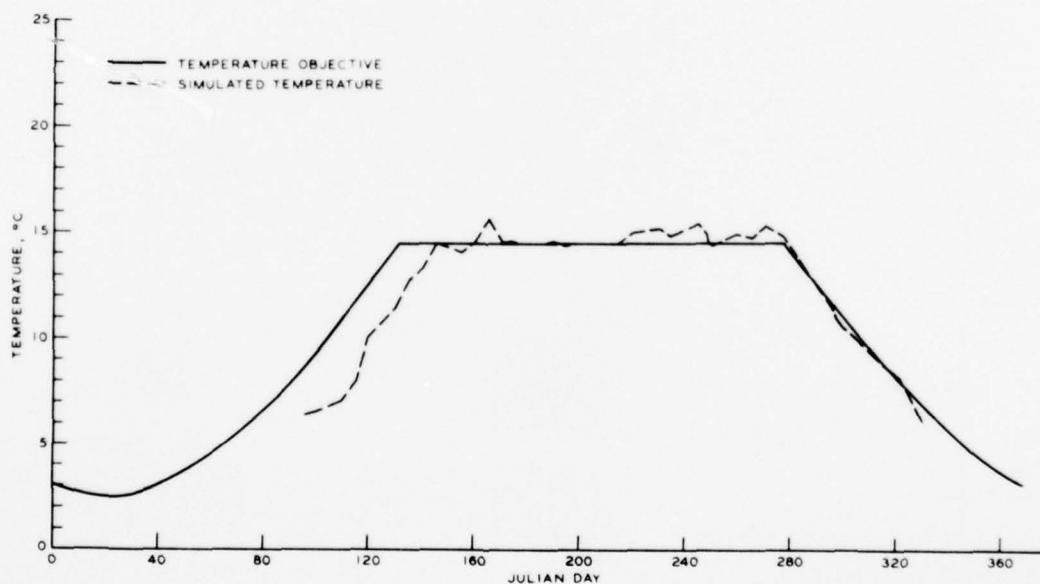


Figure 27. Comparison of temperature objective with simulated release temperature using three ports with blending

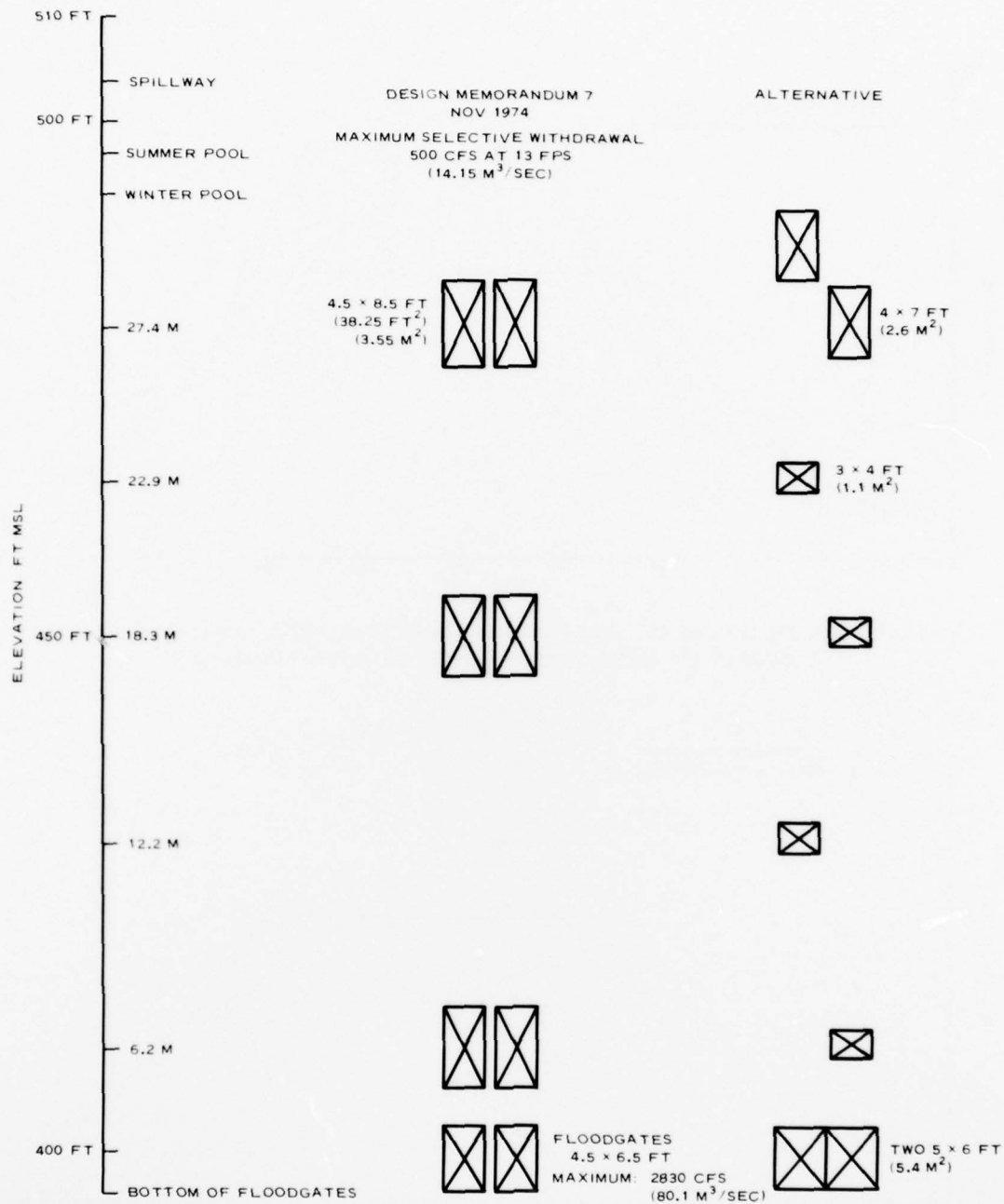


Figure 28. Design Memorandum No. 7 and alternative port and floodgate designs

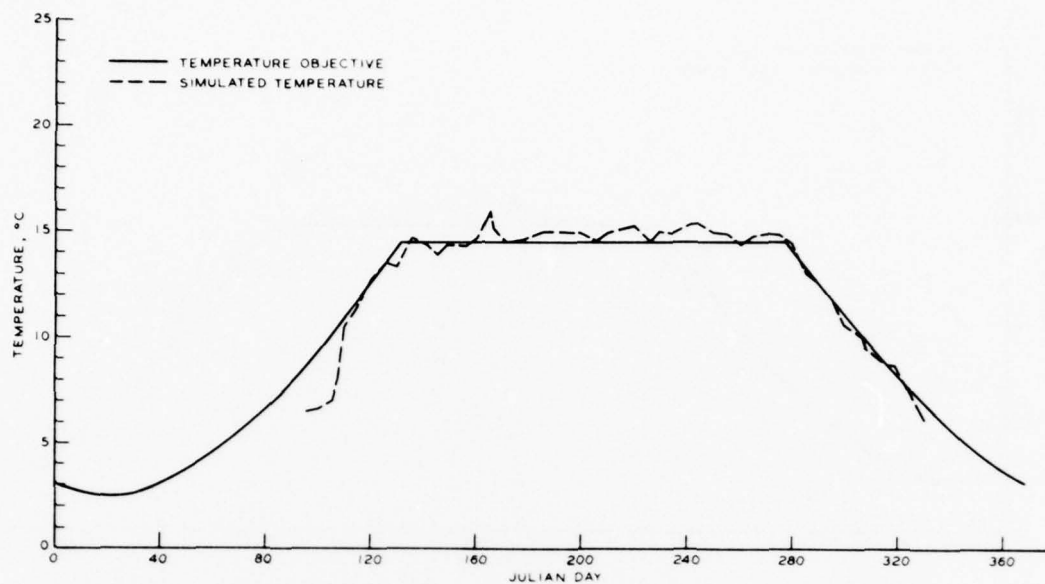


Figure 29. Comparison of temperature objective with 1974 simulated release temperature

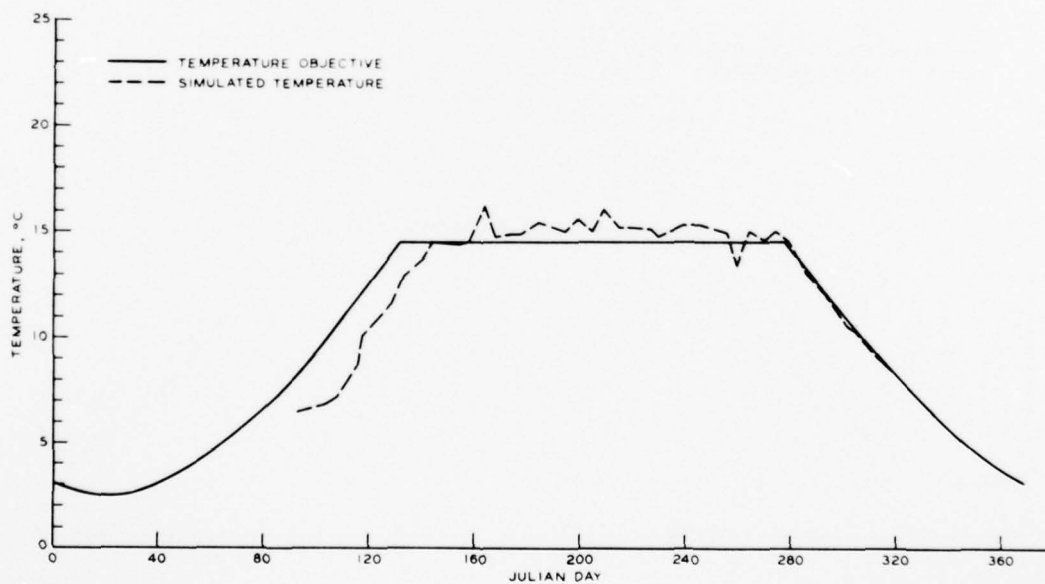


Figure 30. Comparison of temperature objective with 1974 simulated release temperature with top two ports at 27.4 m above streambed

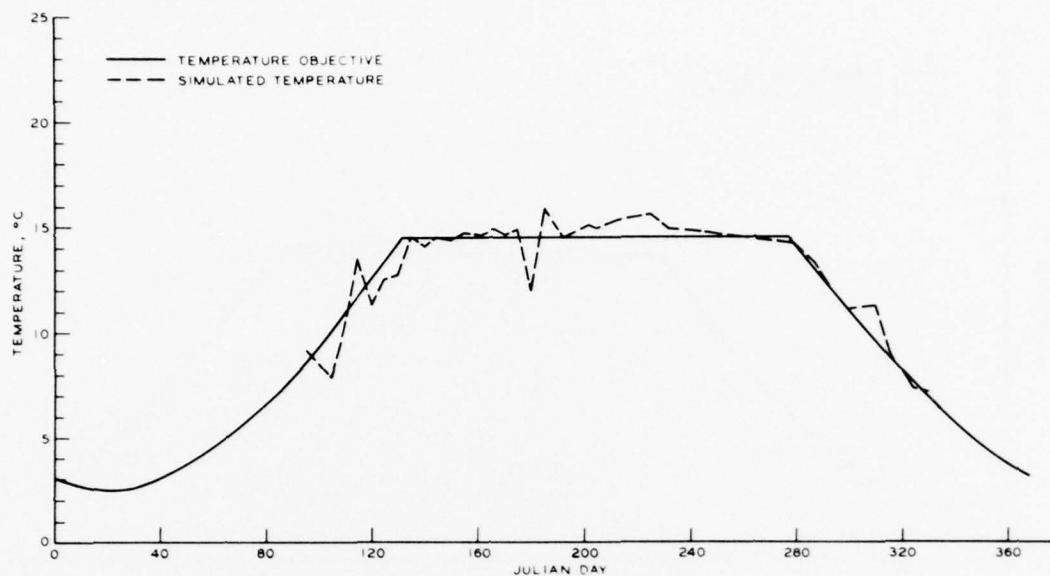


Figure 31. Comparison of temperature objective with 1973 simulated release temperature

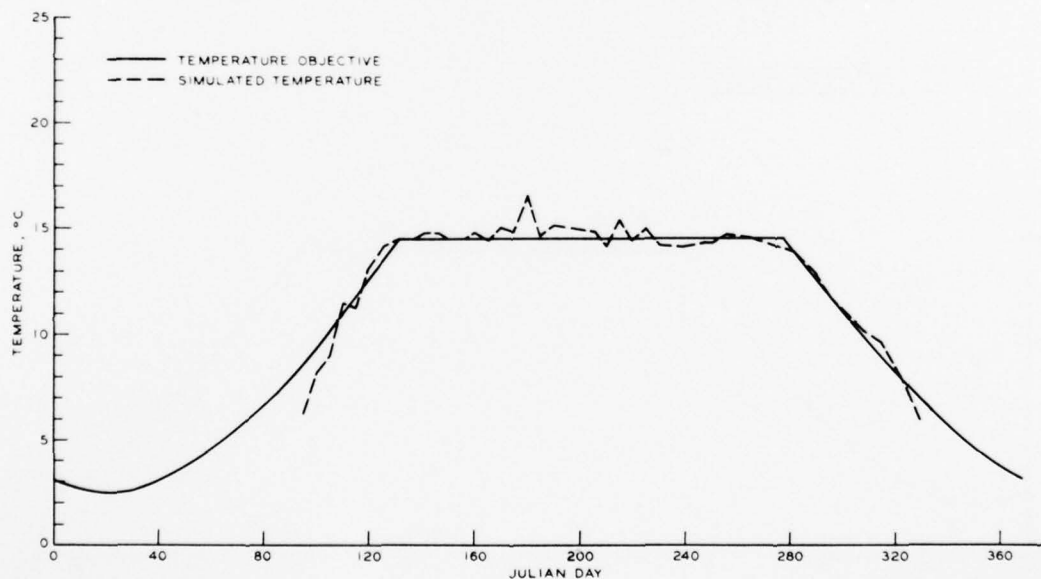


Figure 32. Comparison of temperature objective with 1969 simulated release temperature

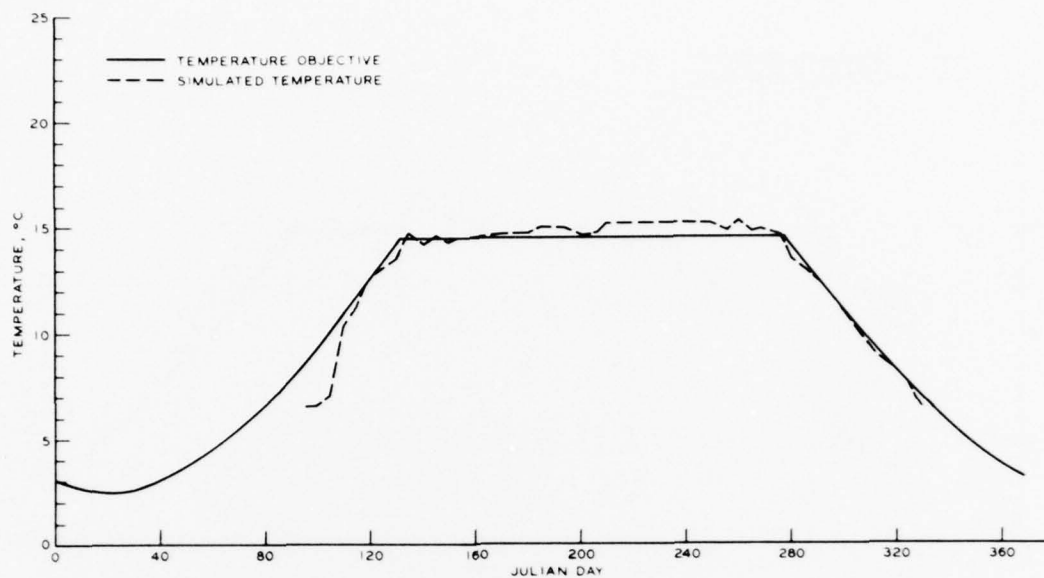


Figure 33. Comparison of temperature objective with 1974 simulated release temperature using the 54-6 outflow

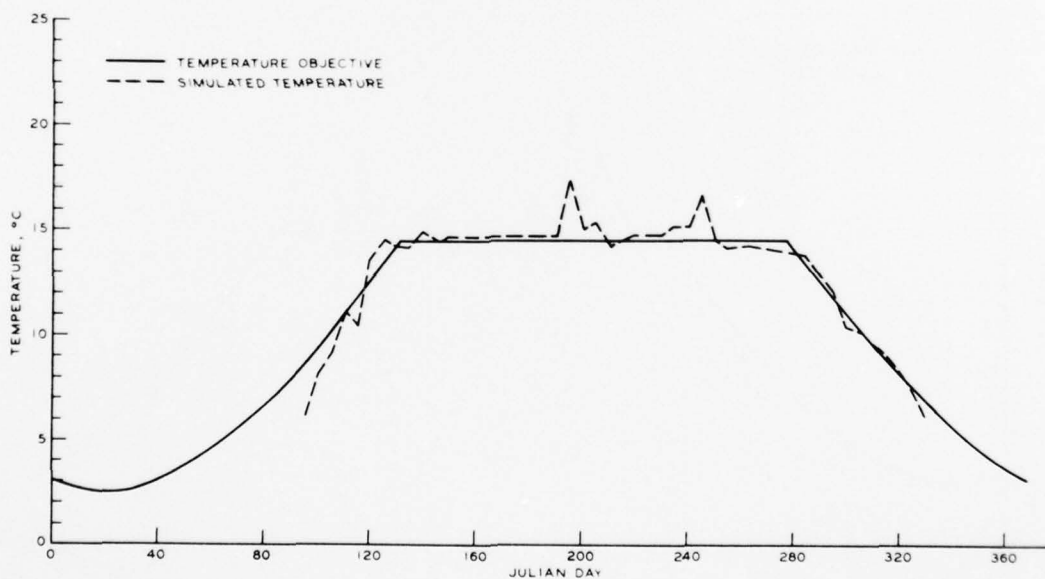


Figure 34. Comparison of temperature objective with 1969 simulated release temperature using the 54-6 outflow

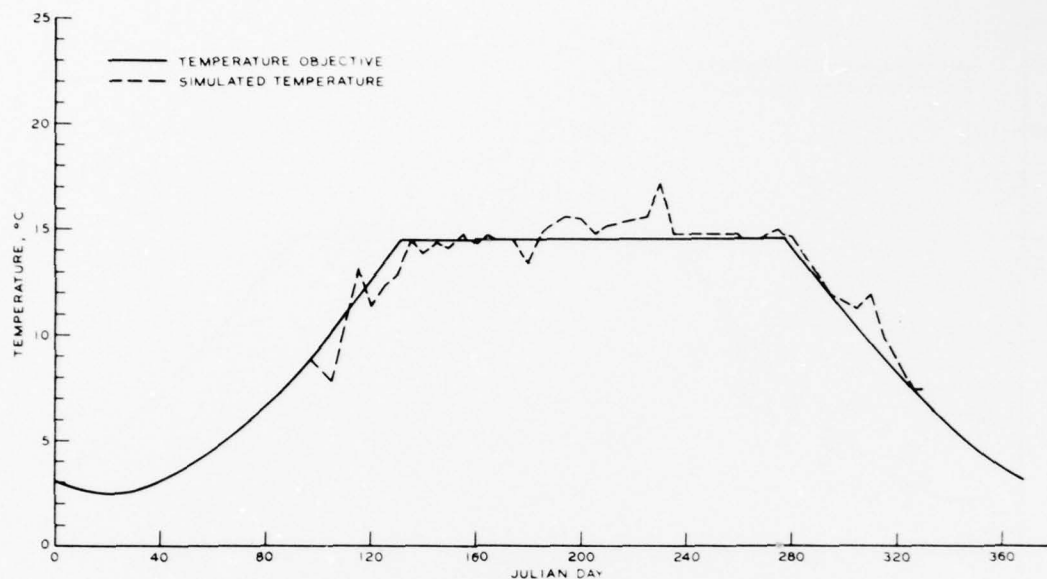


Figure 35. Comparison of temperature objective with 1973 simulated release temperature using the 54-6 outflow

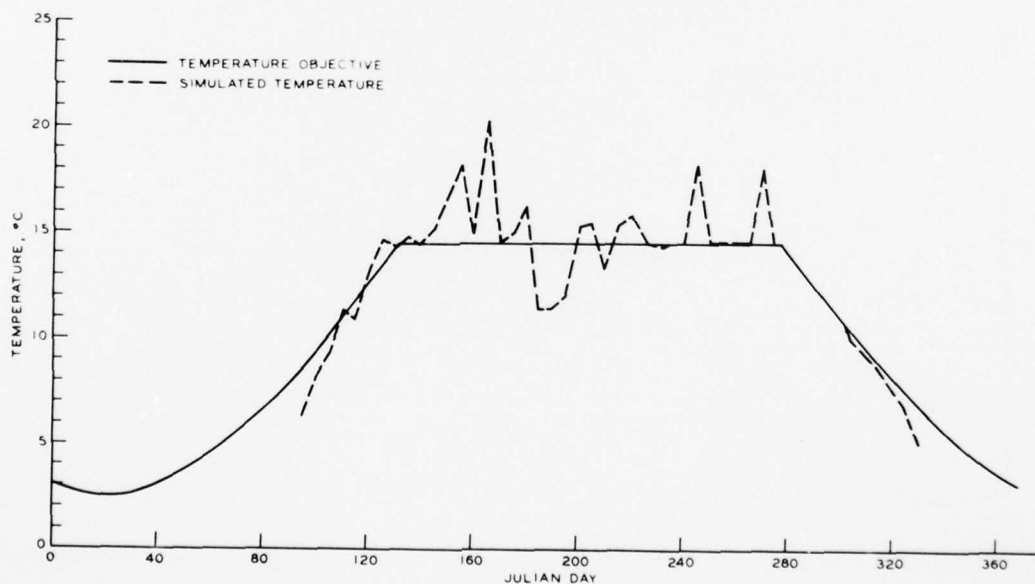


Figure 36. Comparison of temperature objective with 1969 outflow temperatures using 58 and 110 Trenton supplement schedules. Curves were identical for both supplements

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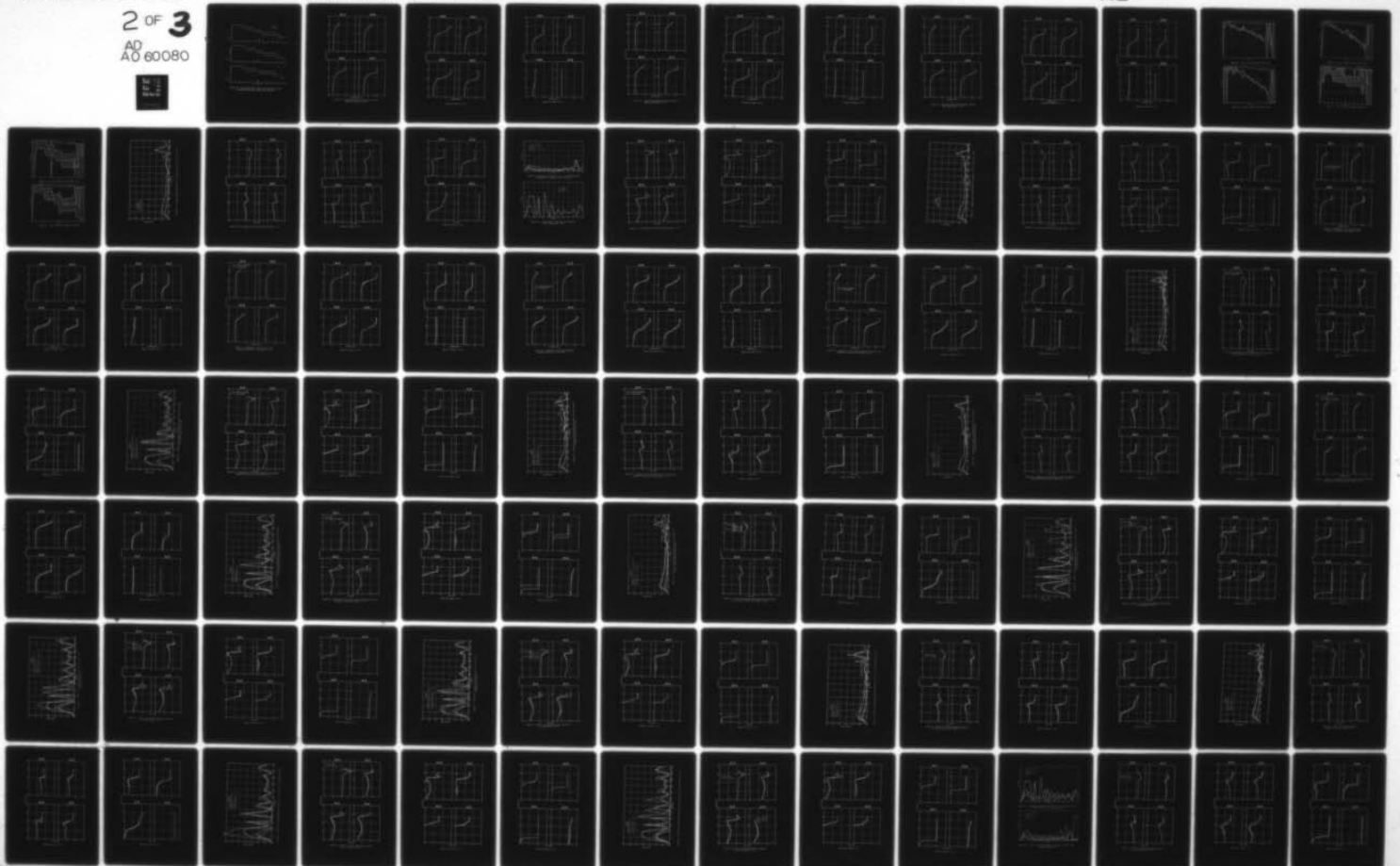
ARMY ENGINEER WATERWAYS EXPERIMENT STATION VICKSBURG MISS F/G 13/2
WATER QUALITY EVALUATION OF PROPOSED TREXLER LAKE, JORDAN CREEK--ETC(U)
AUG 78 D E FORD, K W THORNTON, A S LESSEM

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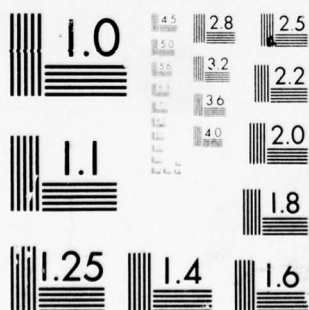
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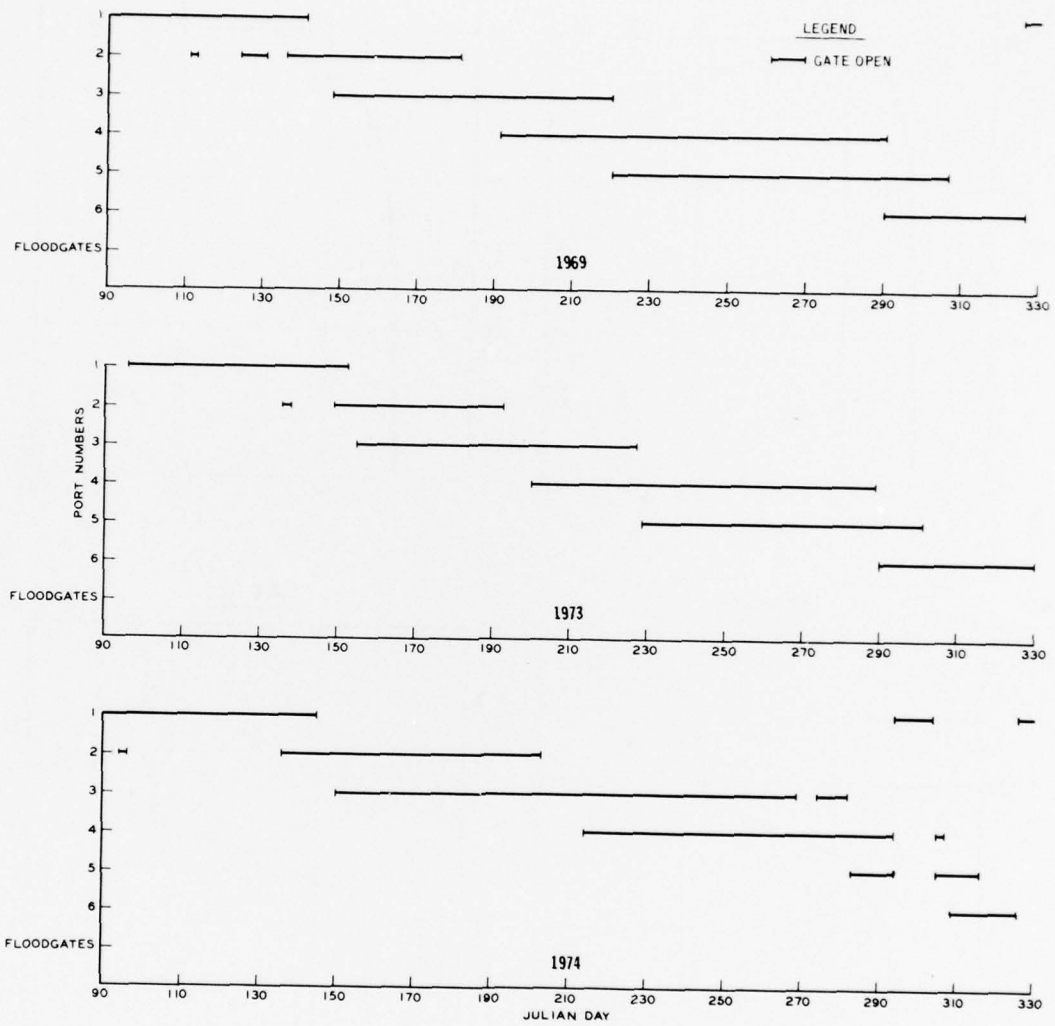


Figure 37. Gate operation for 1969, 1973, and 1974 to meet recommended downstream temperature objectives

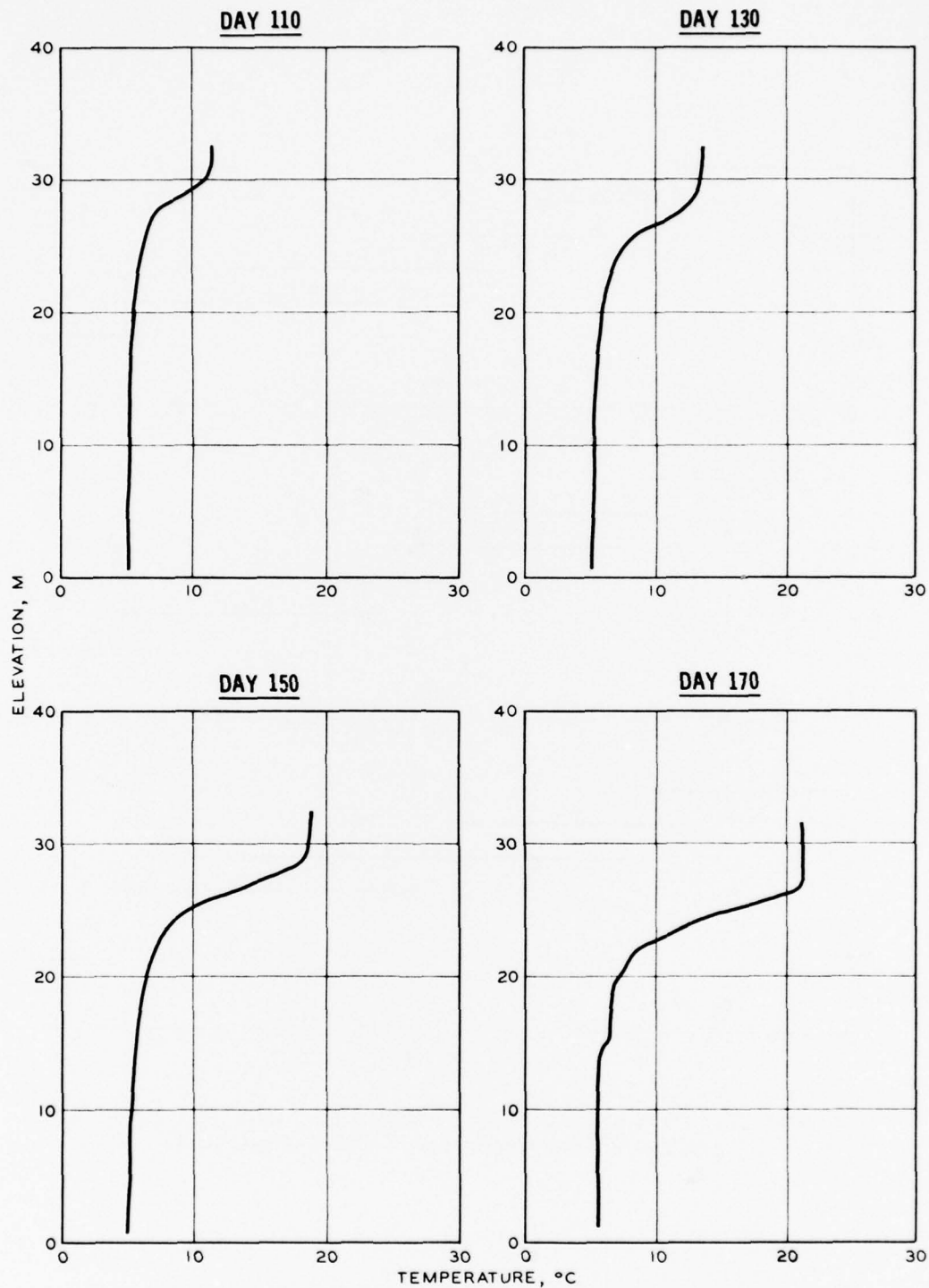


Figure 38. Simulated temperature profiles, Trexler Lake, 1974 (sheet 1 of 3)

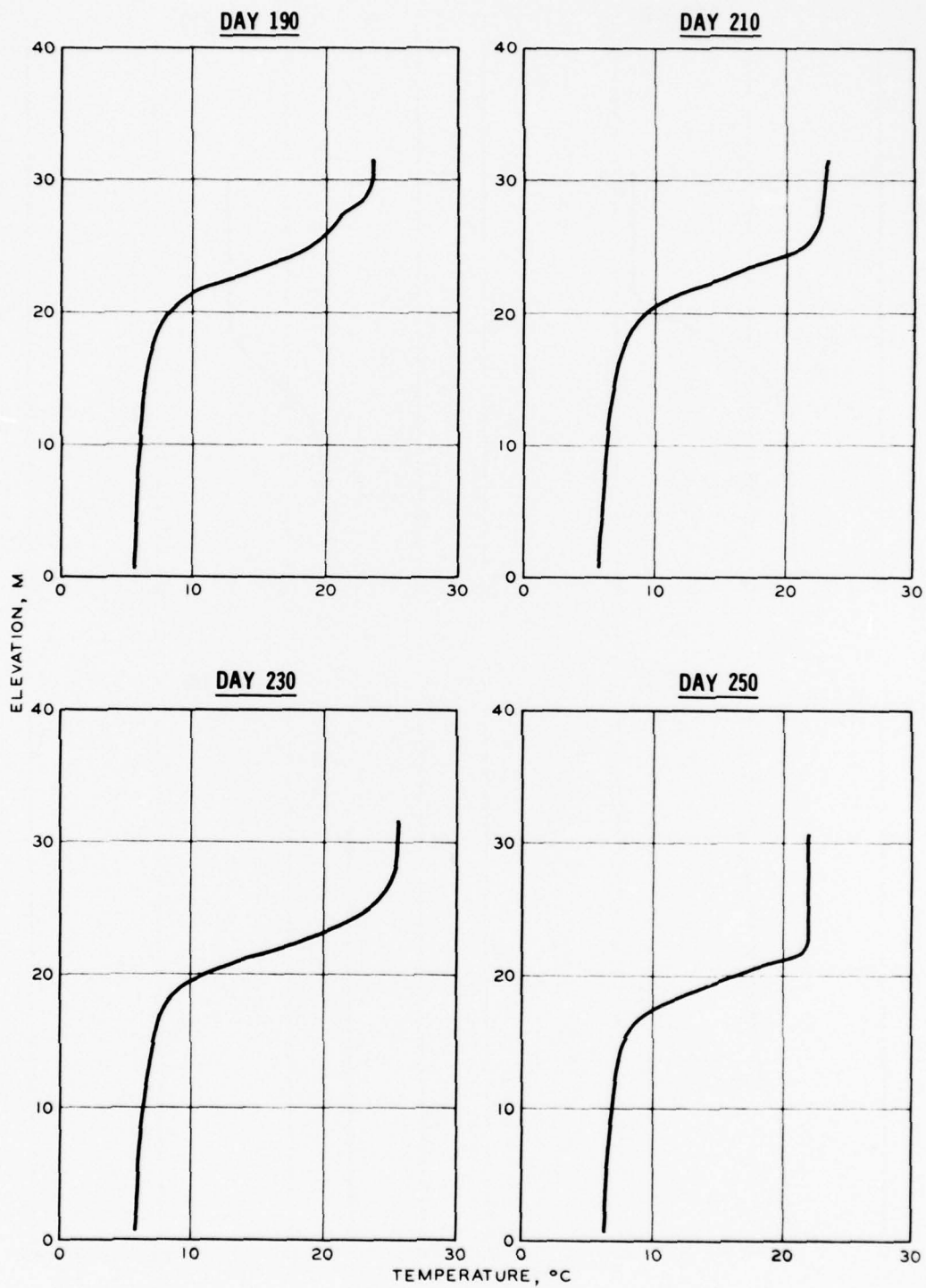


Figure 38 (sheet 2 of 3)

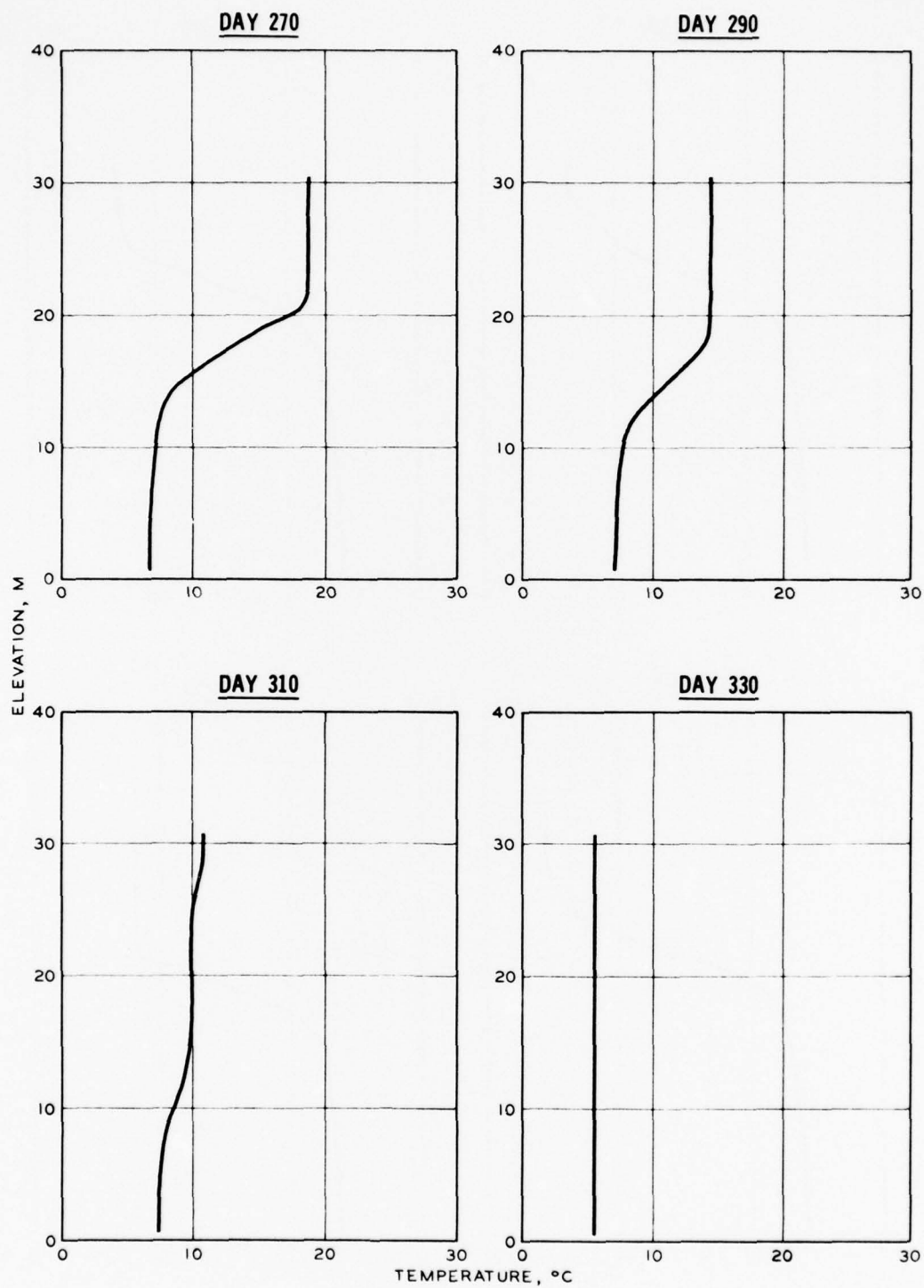


Figure 38 (sheet 3 of 3)

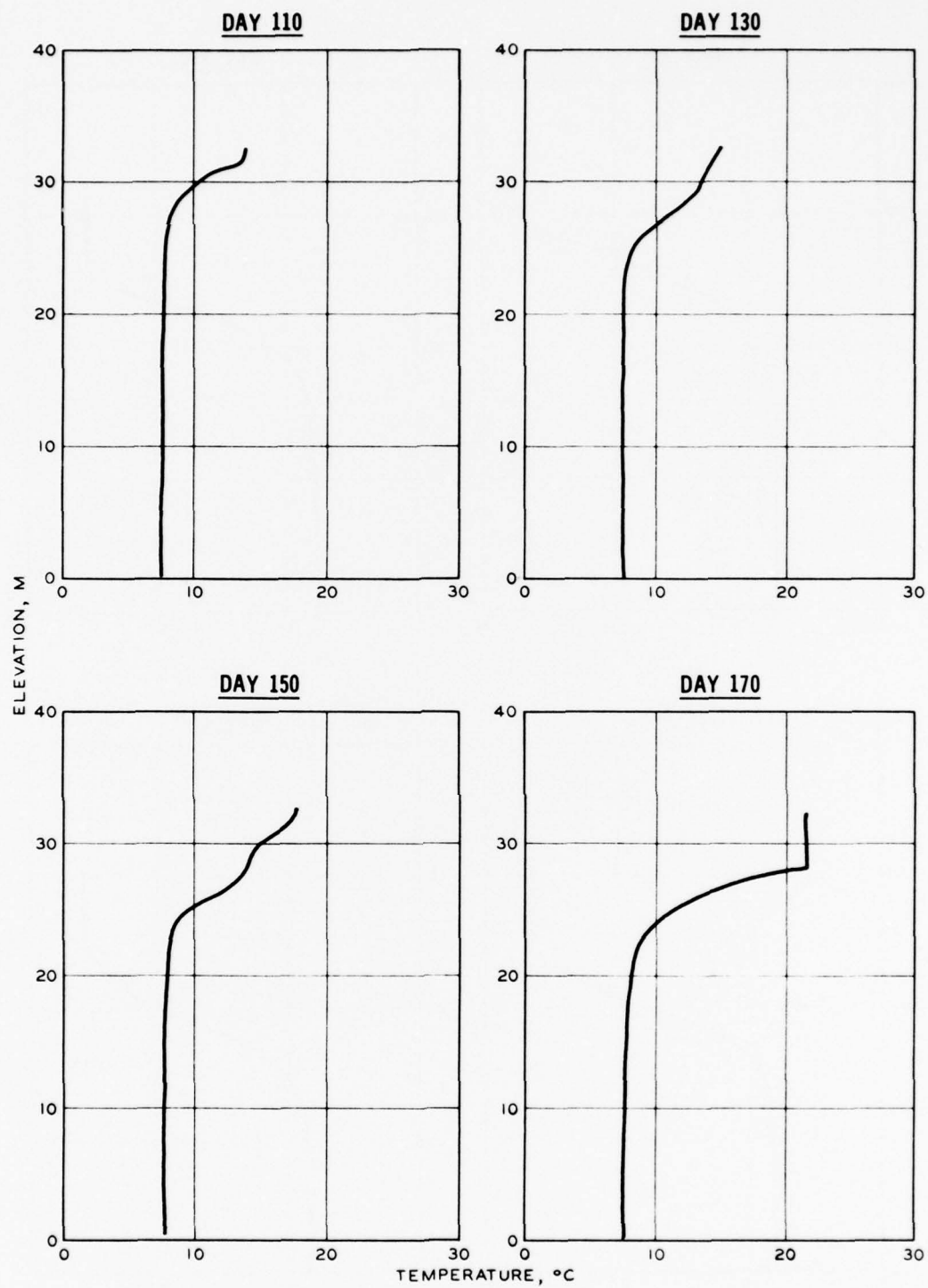


Figure 39. Simulated temperature profiles, Trexler Lake, 1973 (sheet 1 of 3)

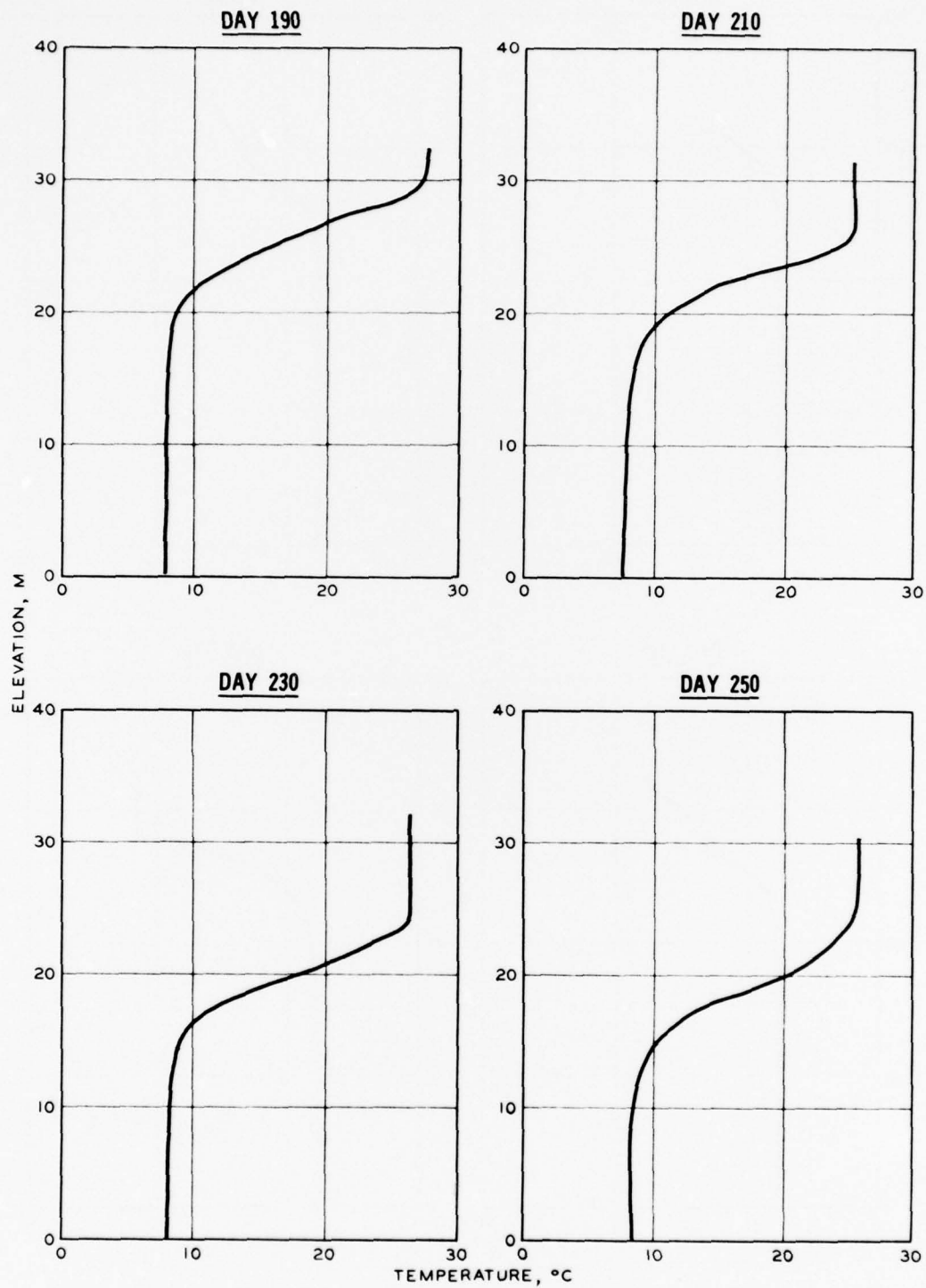


Figure 39 (sheet 2 of 3)

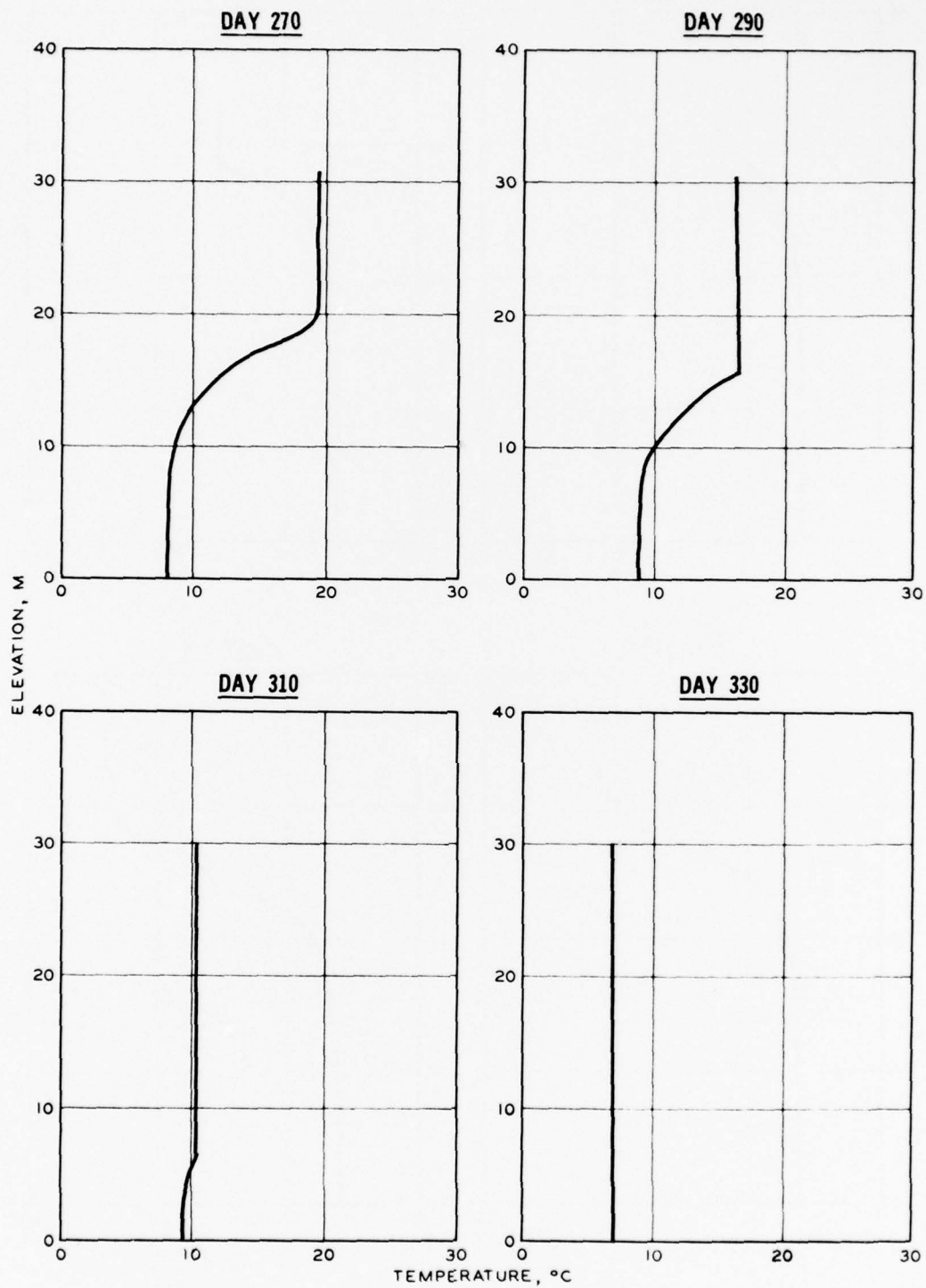


Figure 39 (sheet 3 of 3)

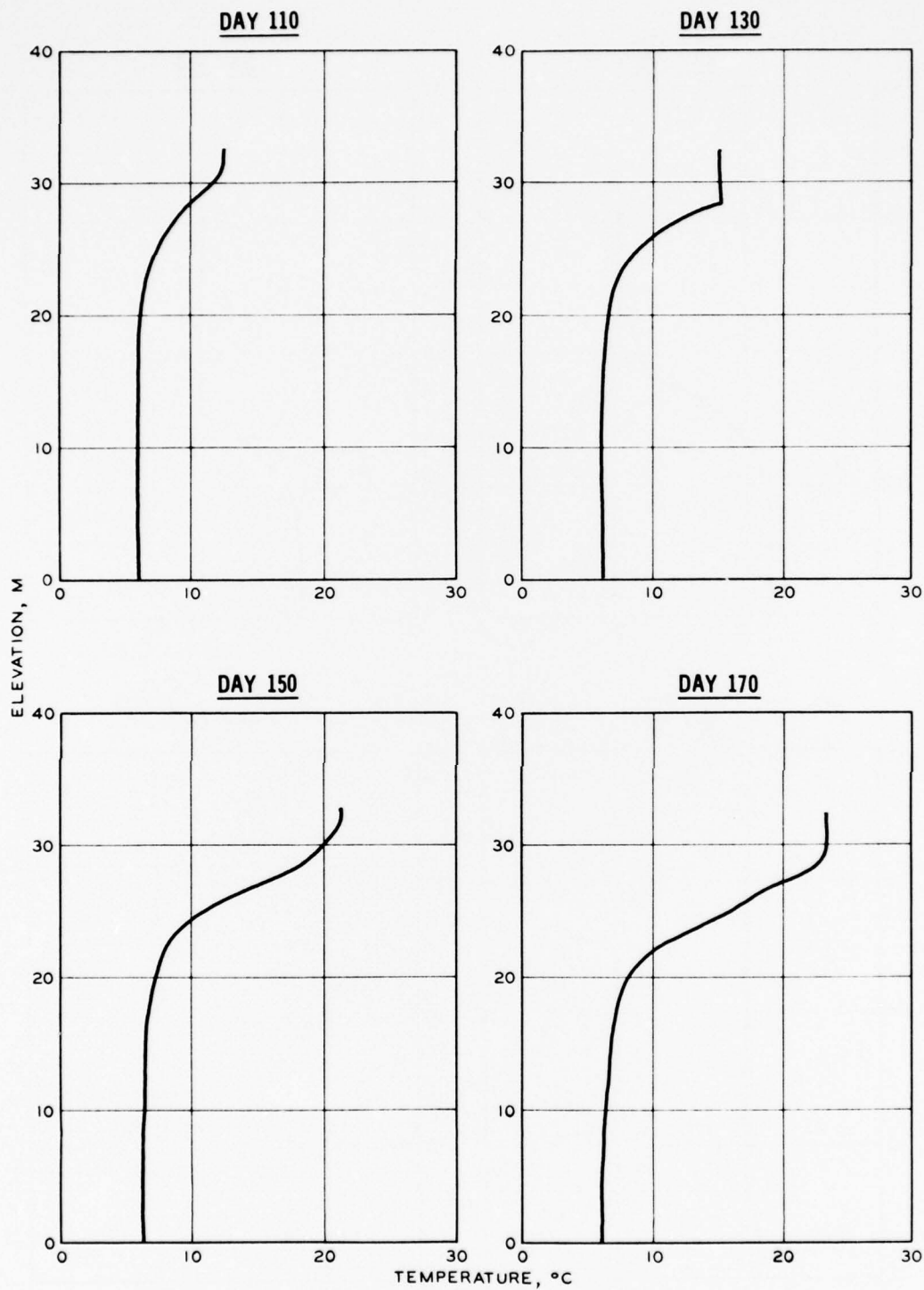


Figure 40. Simulated temperature profiles, Trexler Lake, 1969 (sheet 1 of 3)

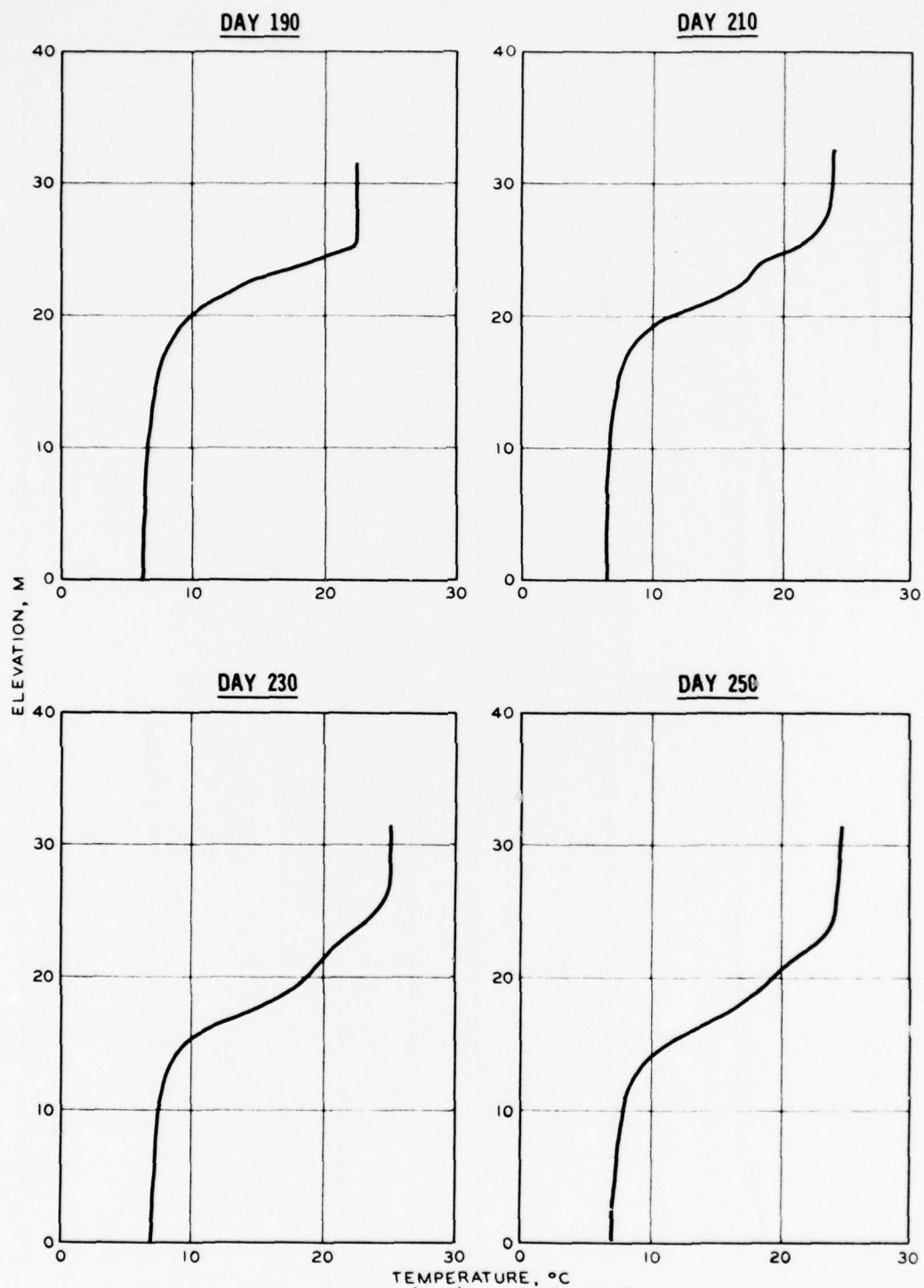


Figure 40 (sheet 2 of 3)

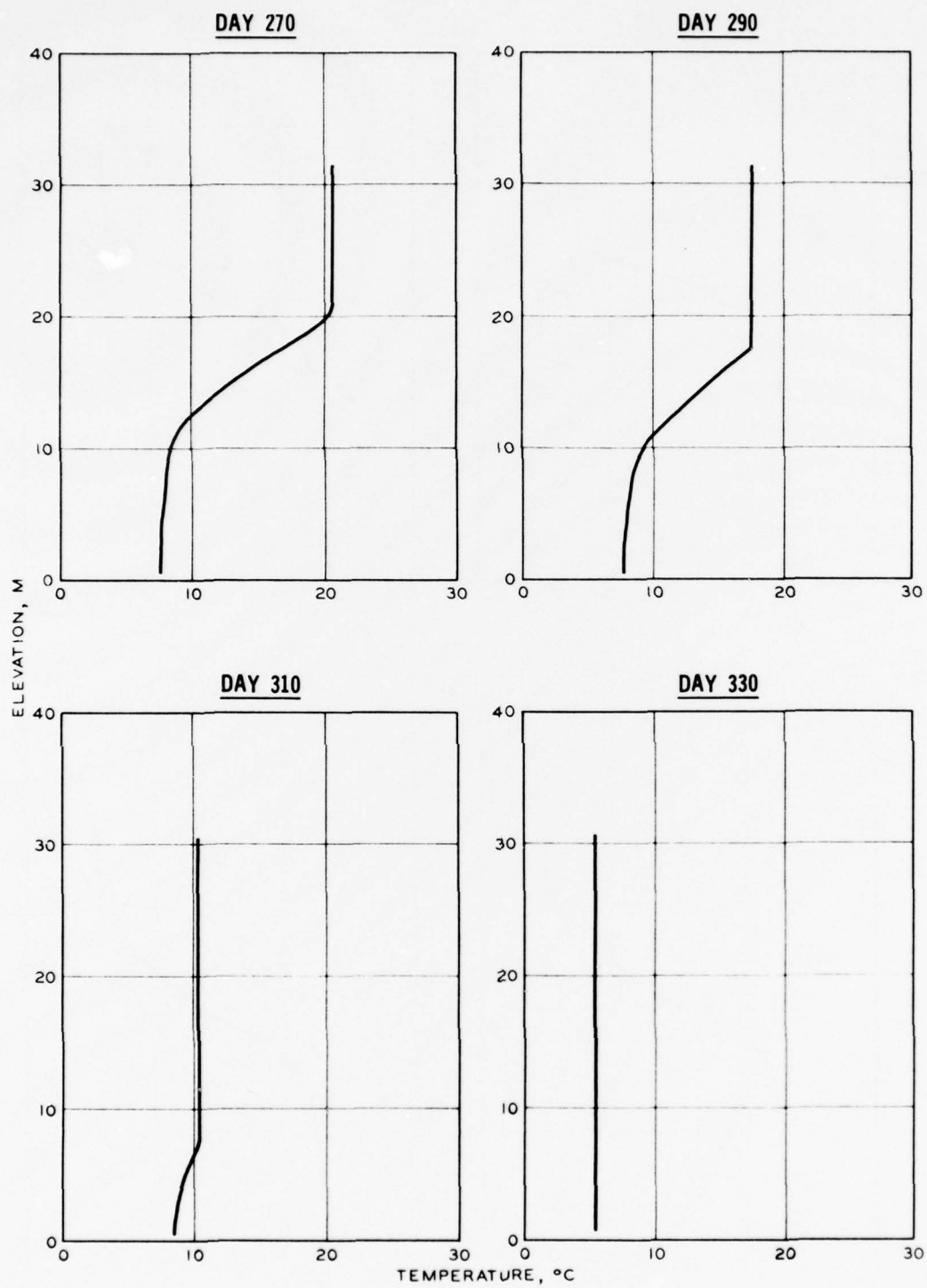


Figure 40 (sheet 3 of 3)

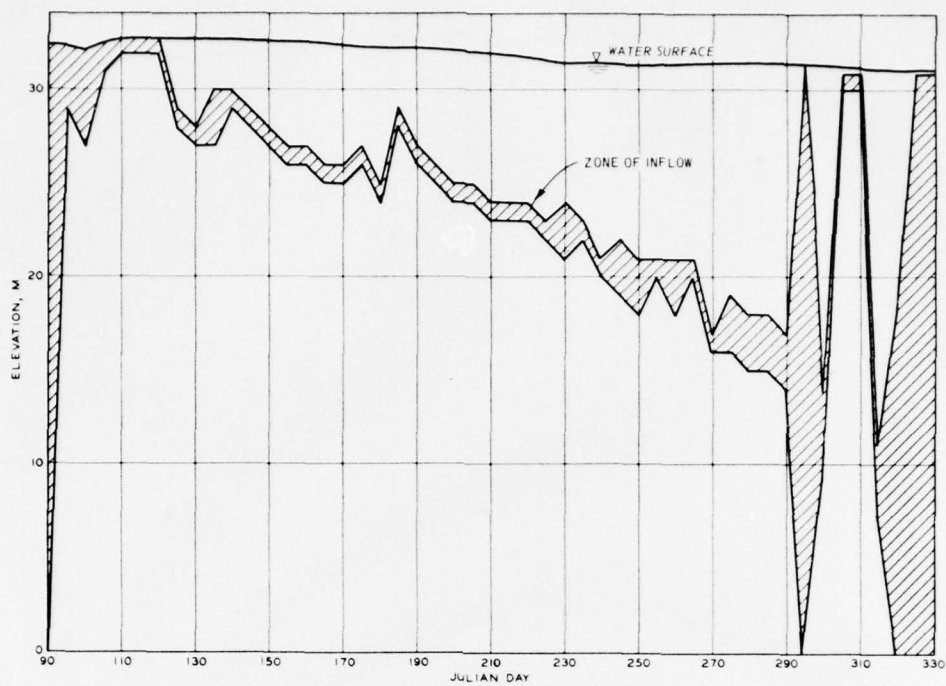


Figure 41. Zone of inflow, Trexler Lake, 1974

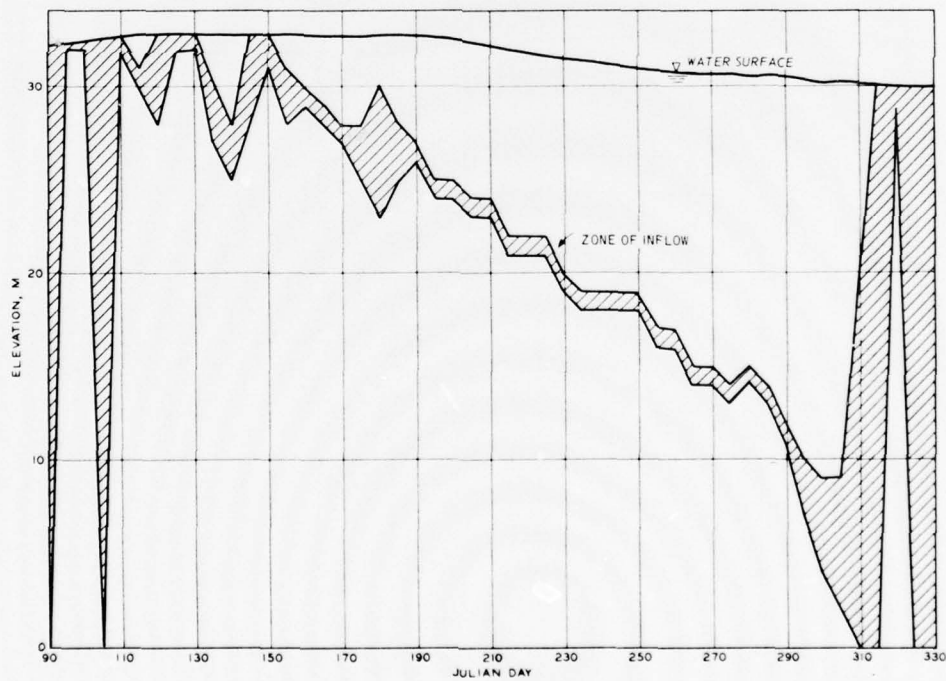


Figure 42. Zone of inflow, Trexler Lake, 1973

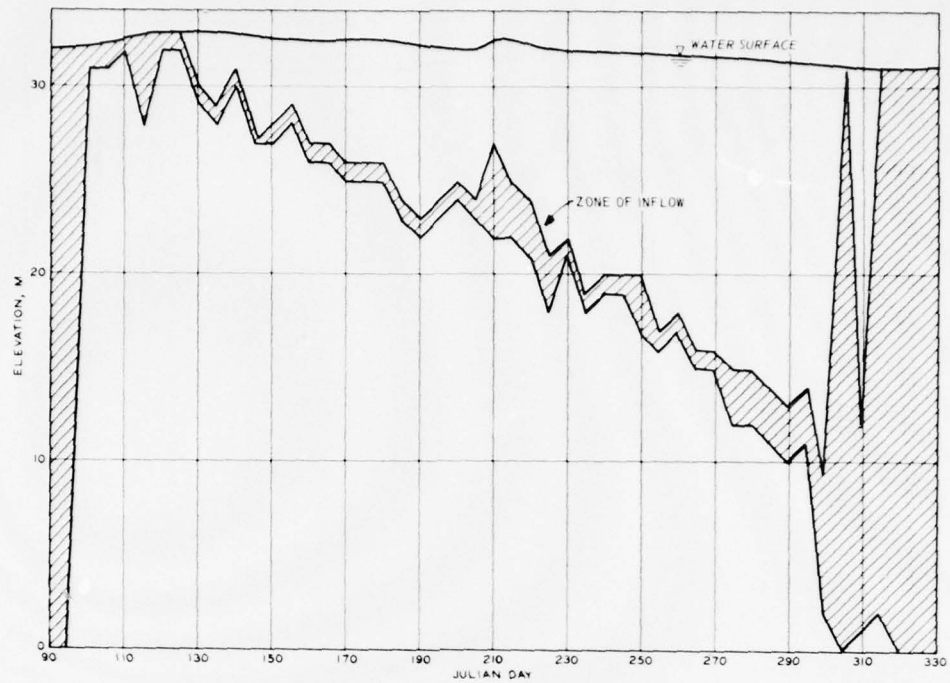


Figure 43. Zone of inflow, Trexler Lake, 1969

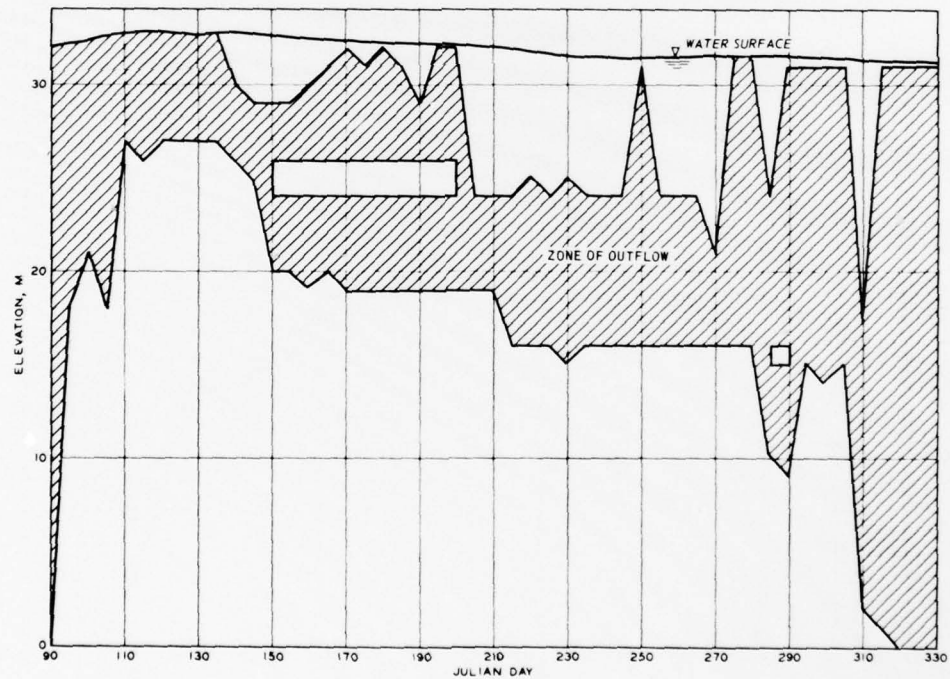


Figure 44. Zone of outflow, Trexler Lake, 1974

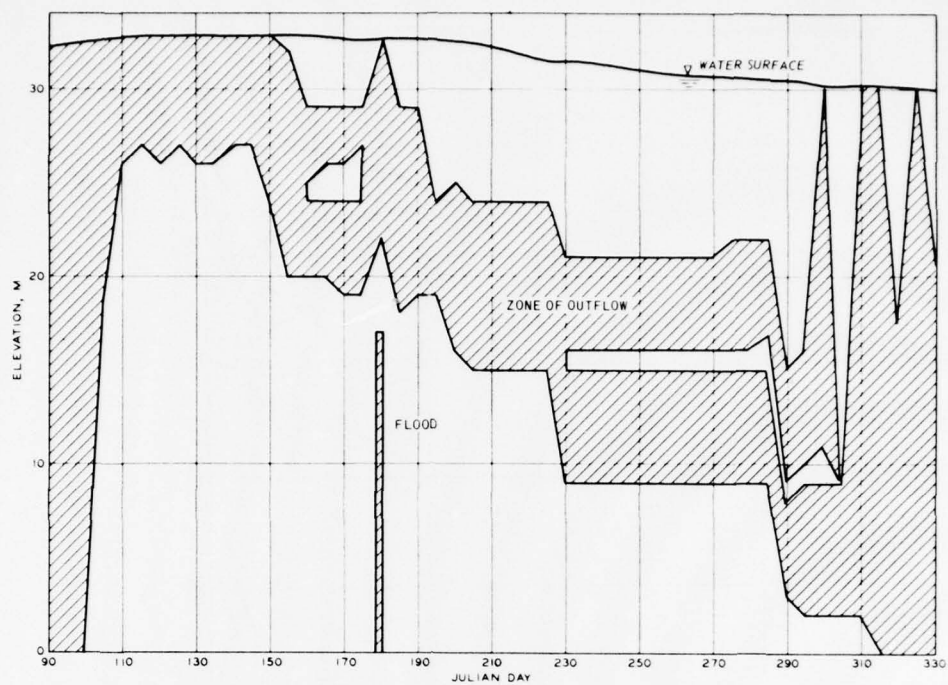


Figure 45. Zone of outflow, Trexler Lake, 1973

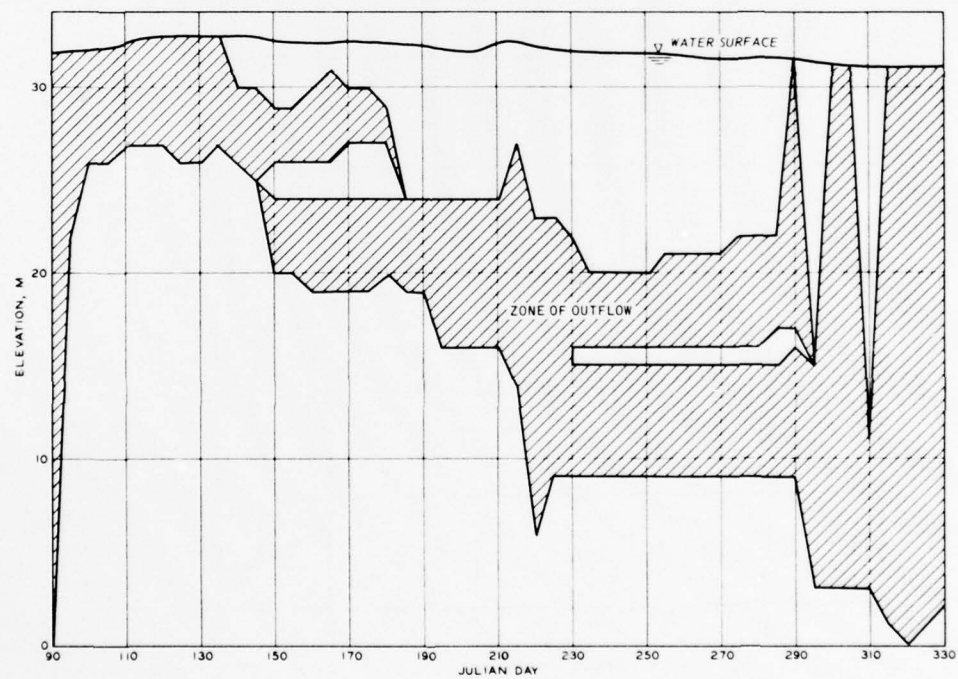


Figure 46. Zone of outflow, Trexler Lake, 1969

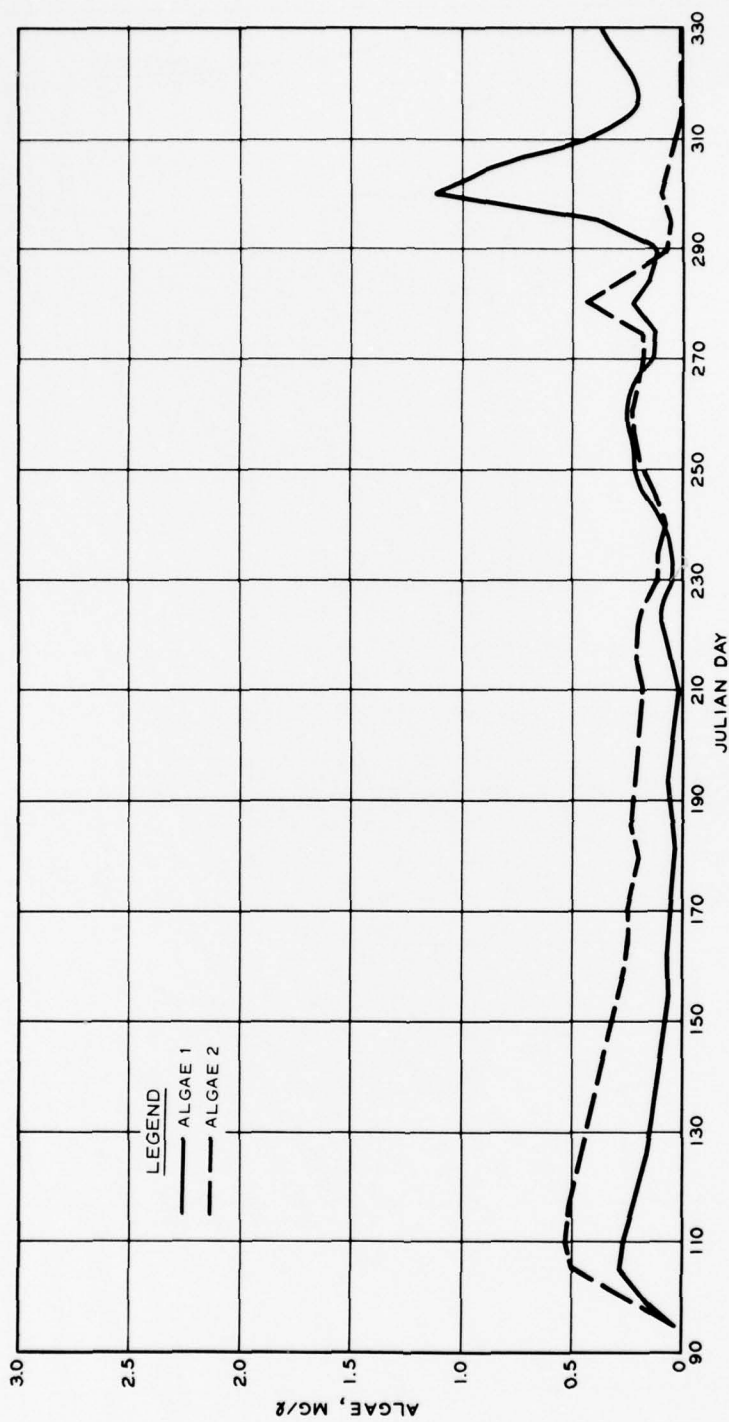


Figure 47. Average algae concentrations in euphotic zone for base case, 1974

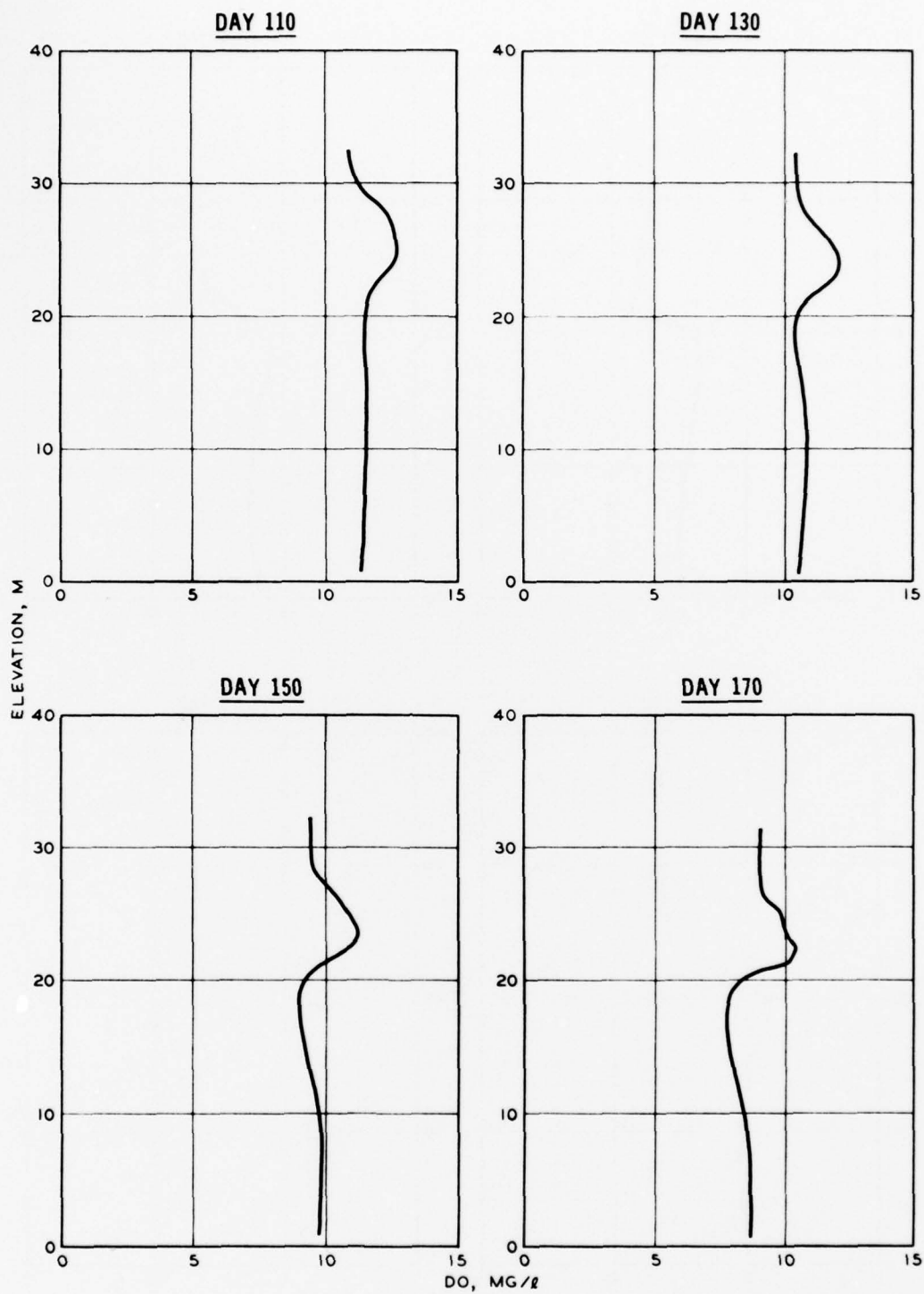


Figure 48. DO profiles for base case, 1974 (sheet 1 of 3)

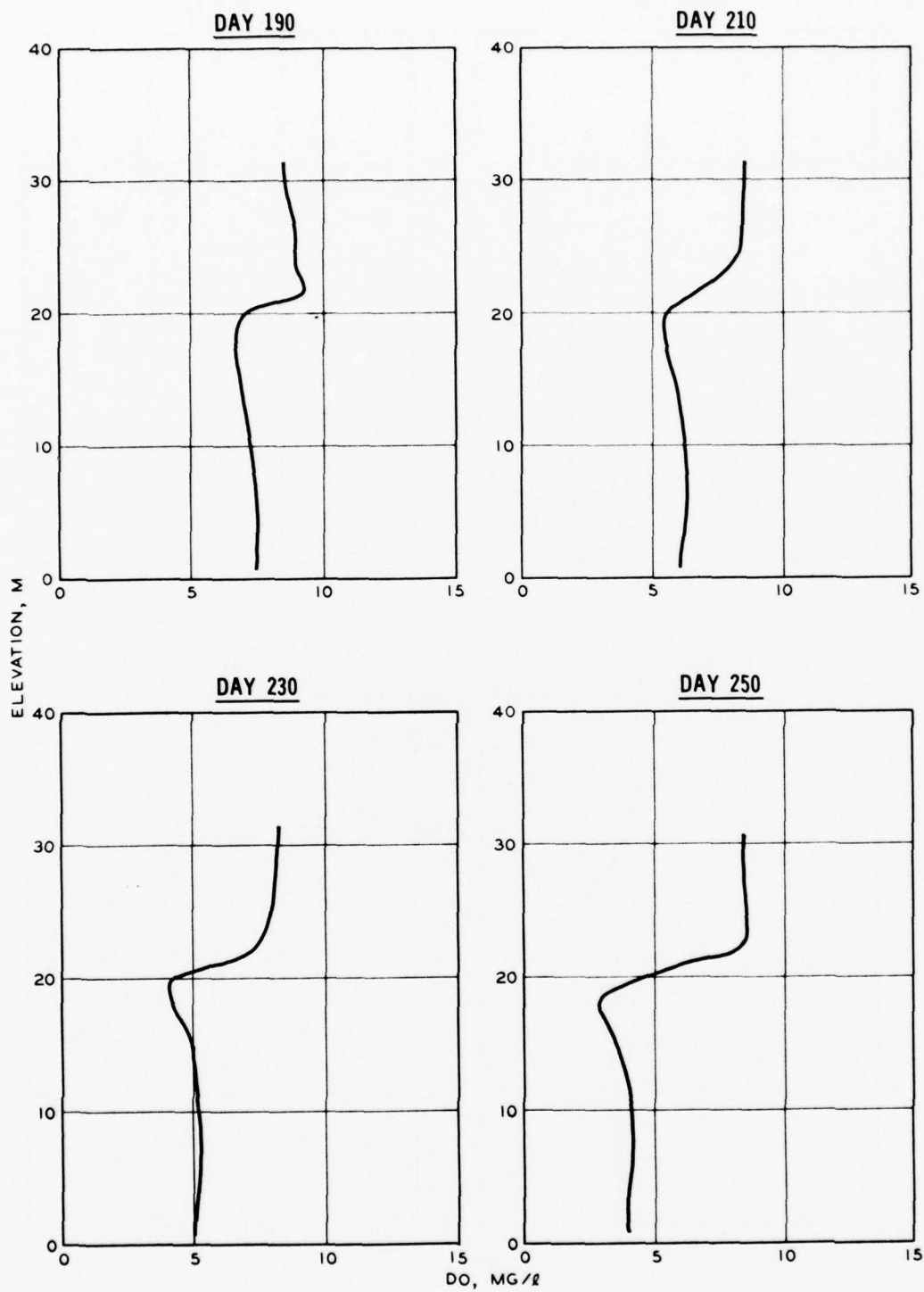


Figure 48 (sheet 2 of 3)

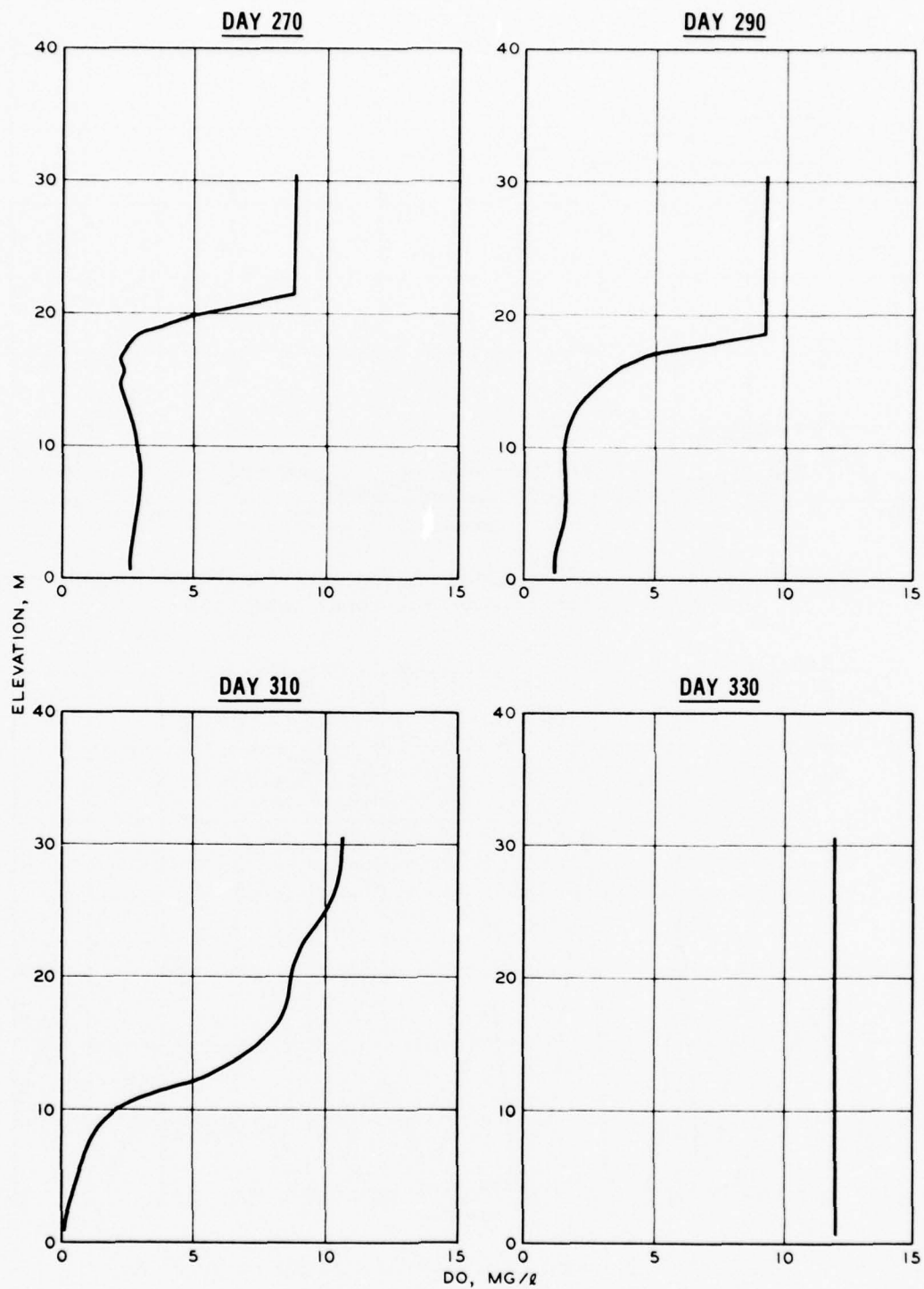


Figure 48 (sheet 3 of 3)

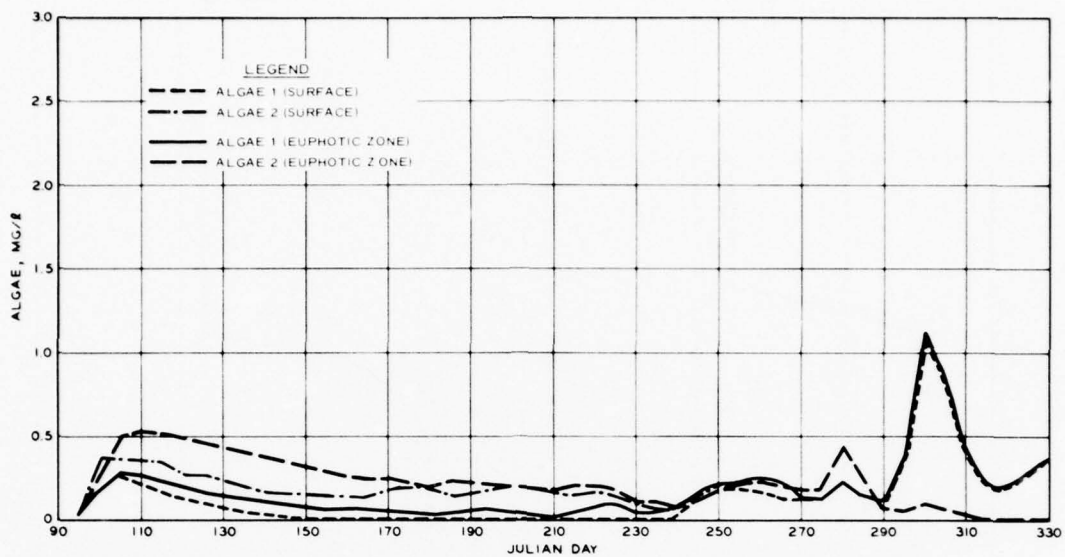


Figure 49. Comparison of algae in top 1 m and average algae concentration in euphotic zone, base case

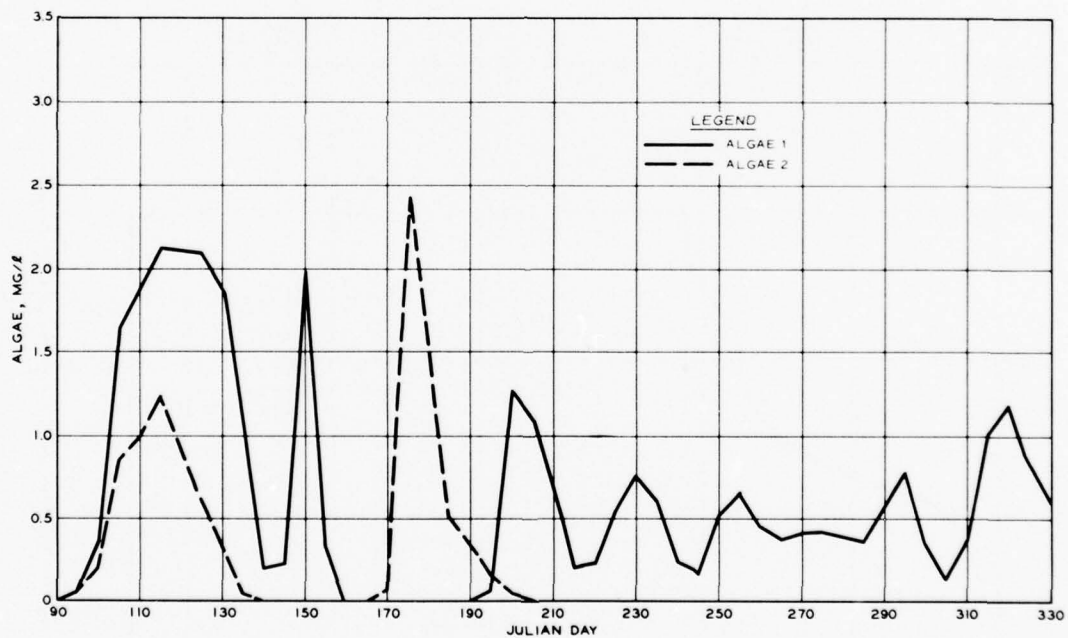


Figure 50. Average algae concentrations in euphotic zone for base case, 1973

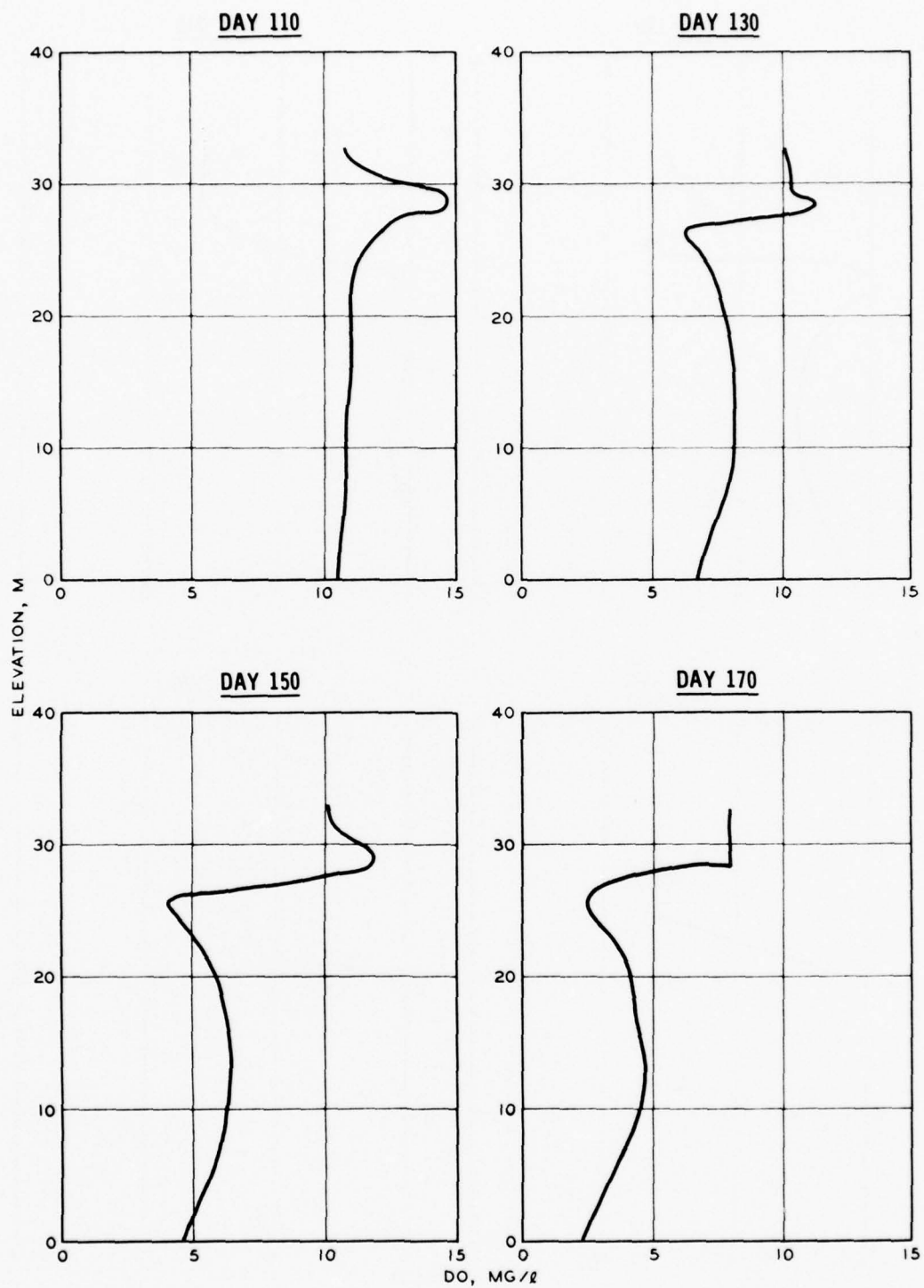


Figure 51. DO profiles for base case, 1973 (sheet 1 of 3)

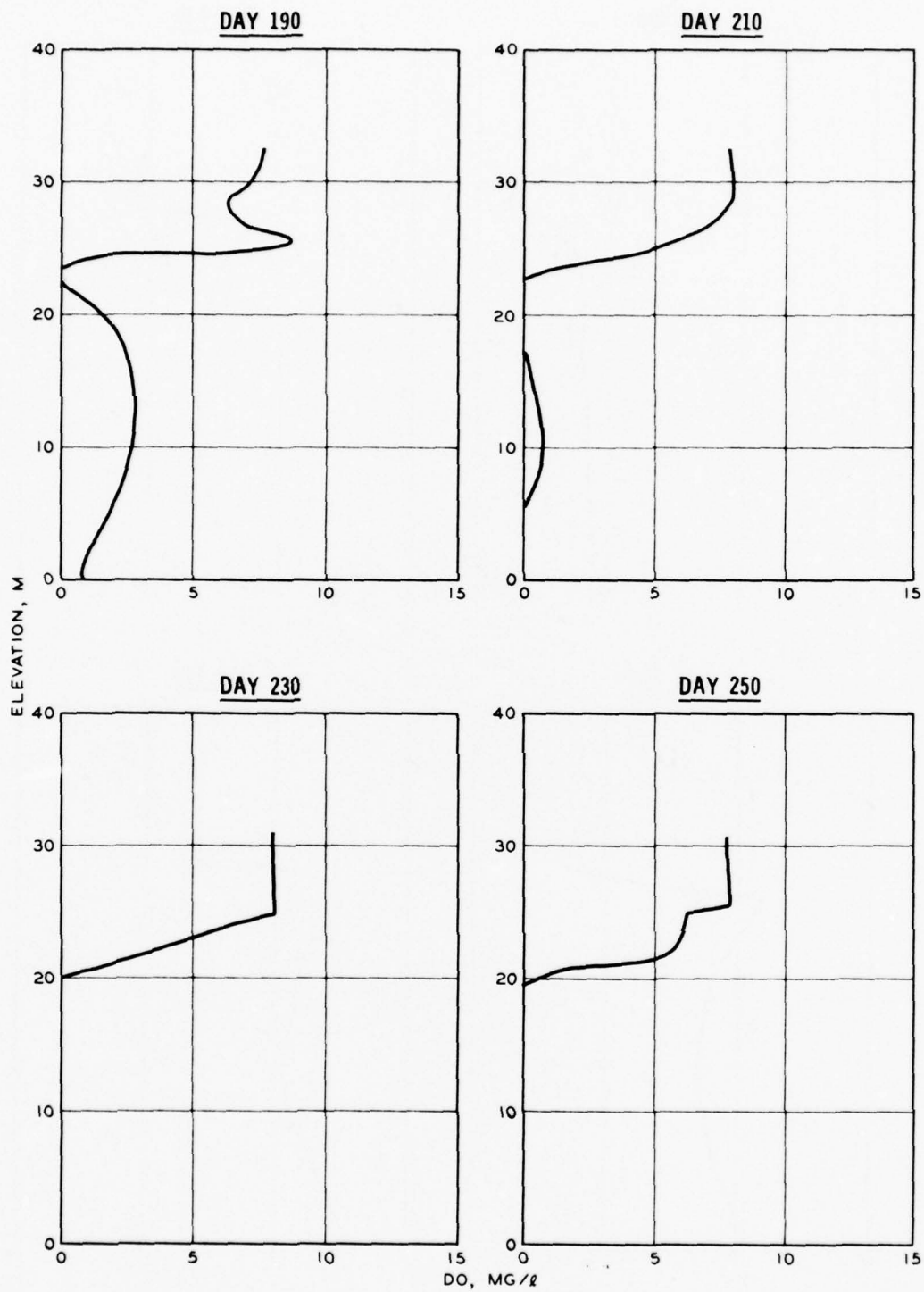


Figure 51 (sheet 2 of 3)

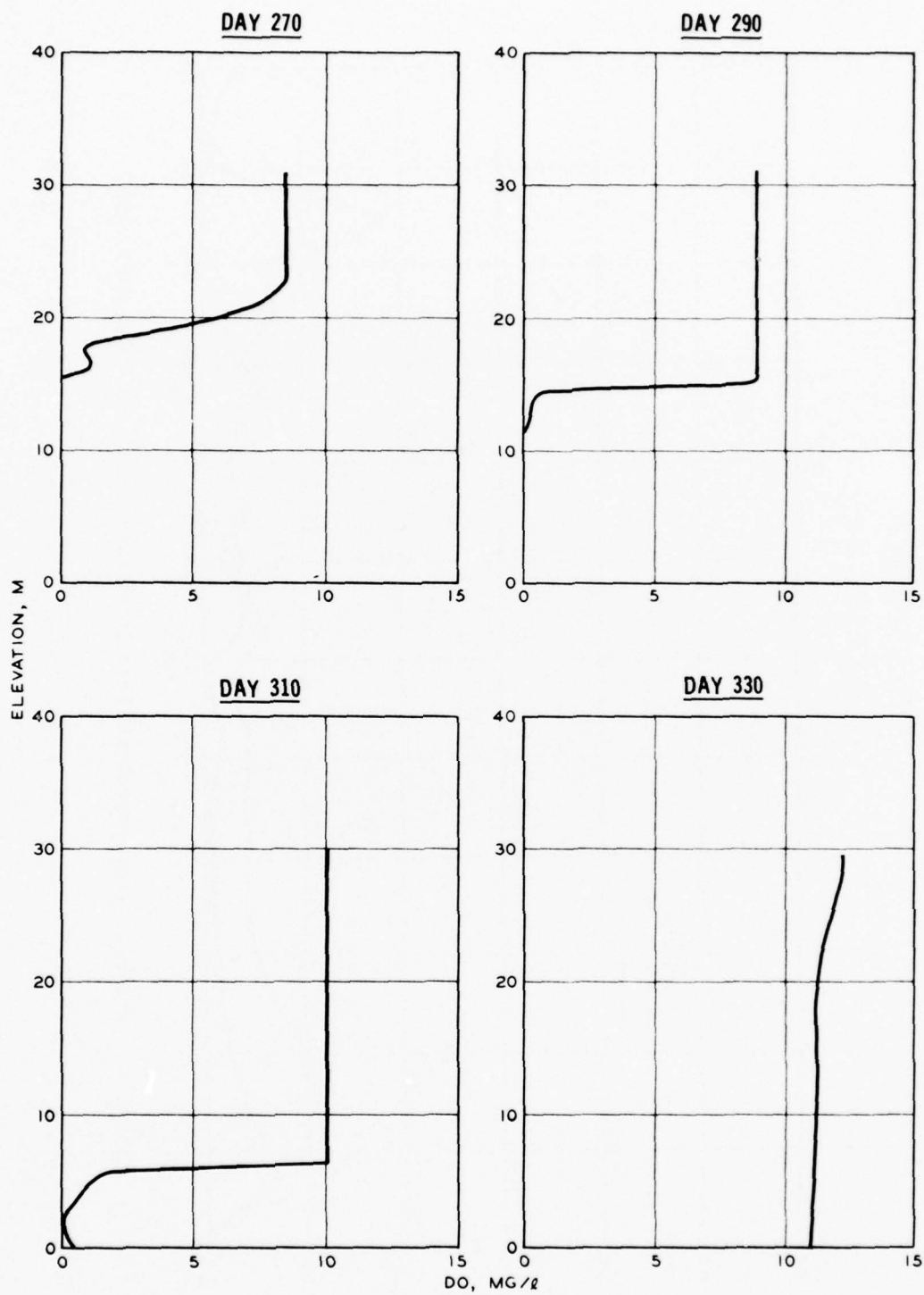


Figure 51 (sheet 3 of 3)

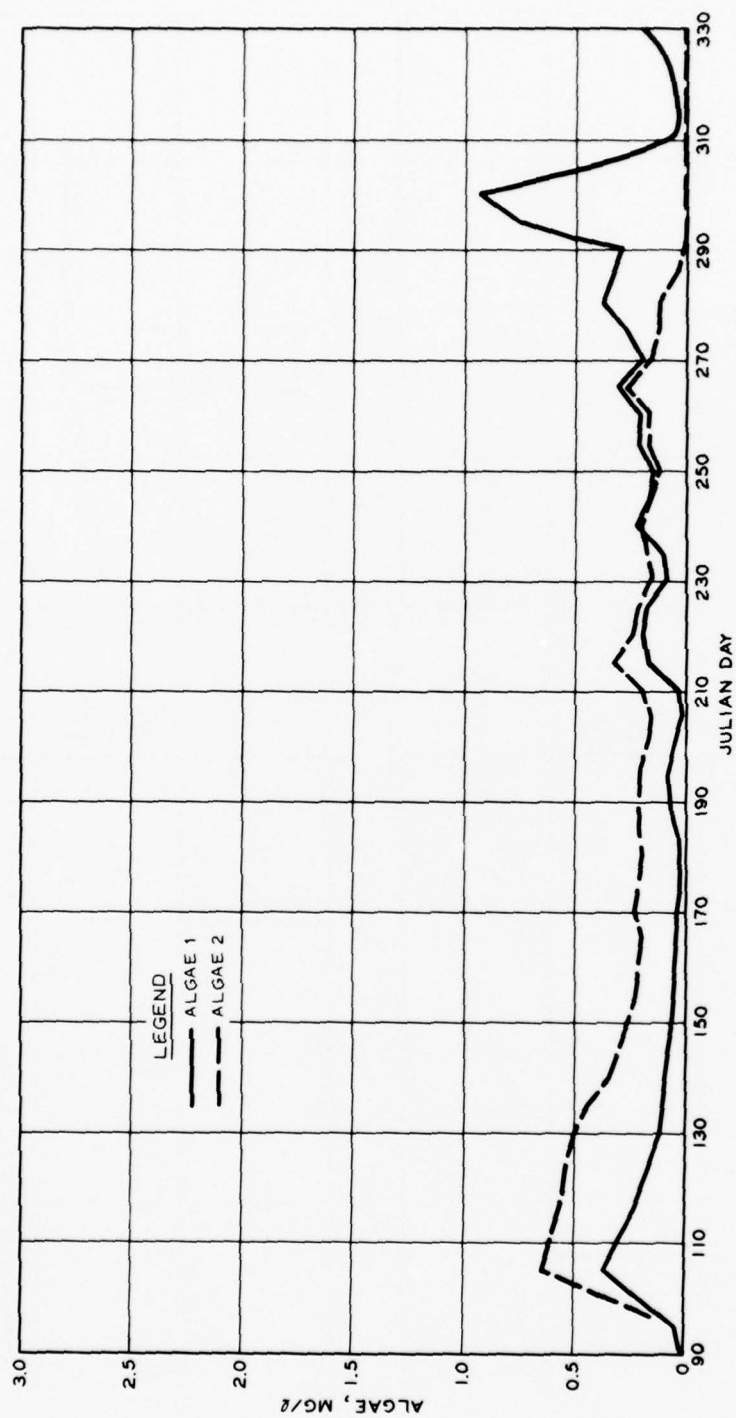


Figure 52. Average algae concentrations in euphotic zone for base case, 1969

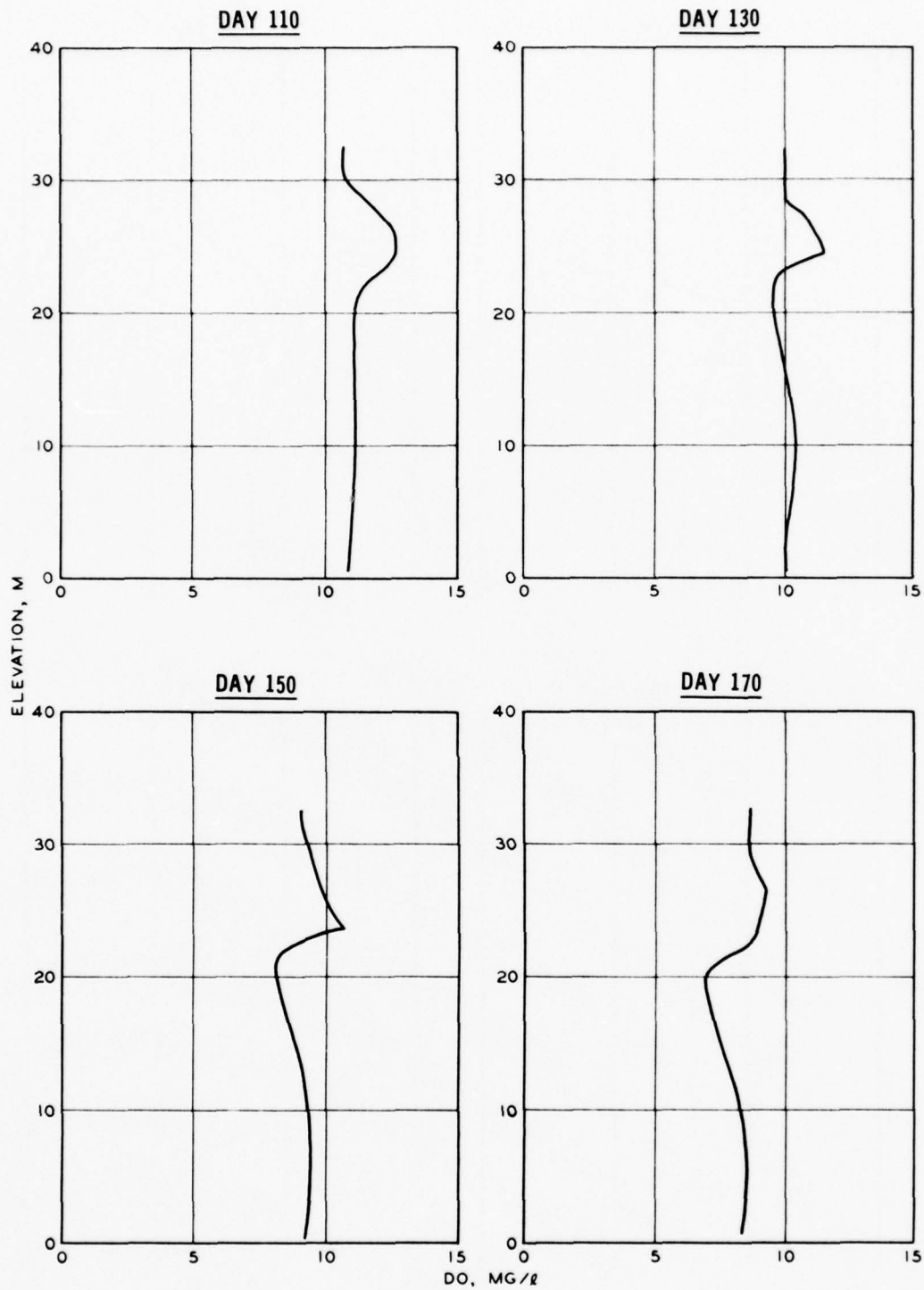


Figure 53. DO profiles for base case, 1969 (sheet 1 of 3)

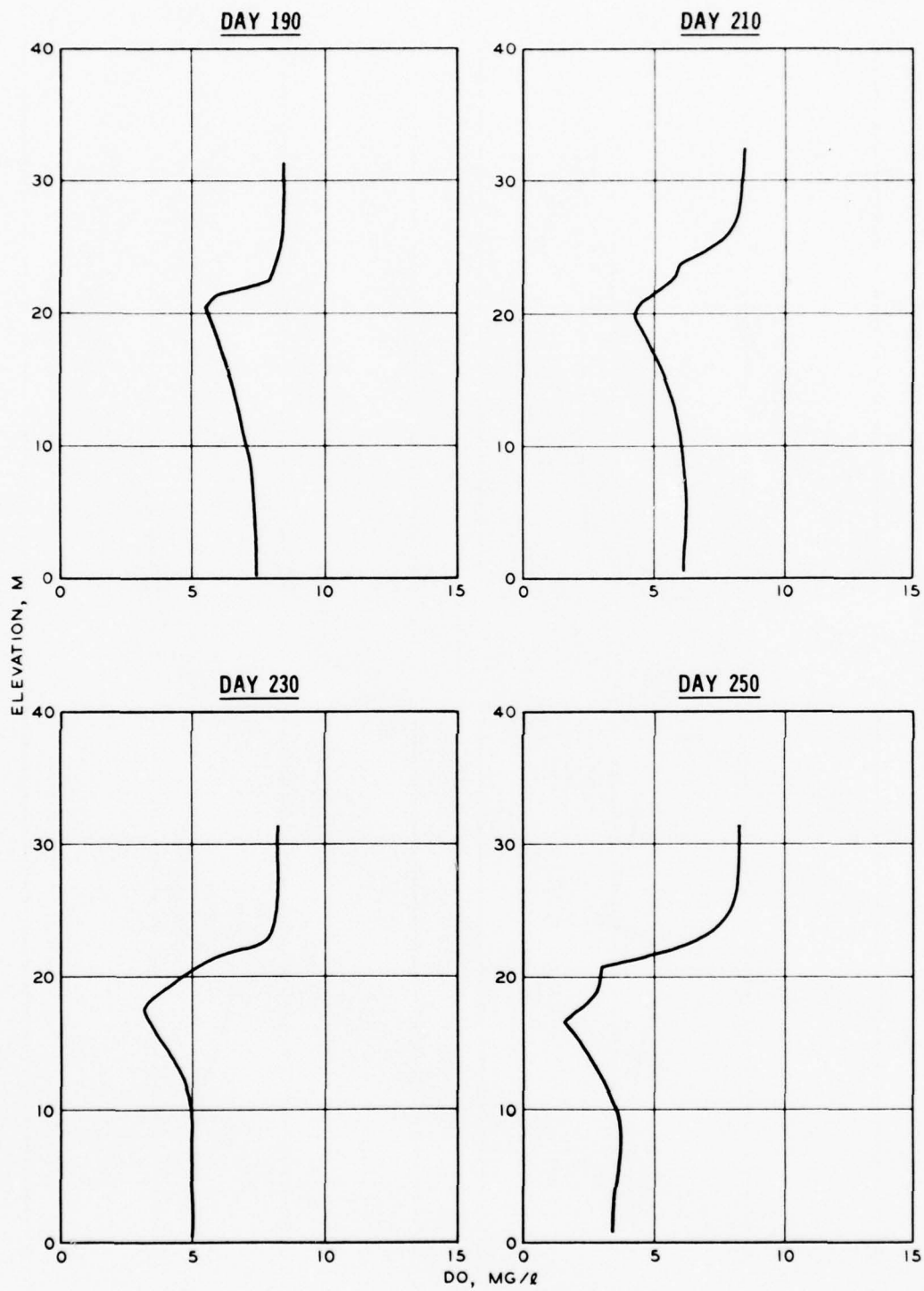


Figure 53 (sheet 2 of 3)

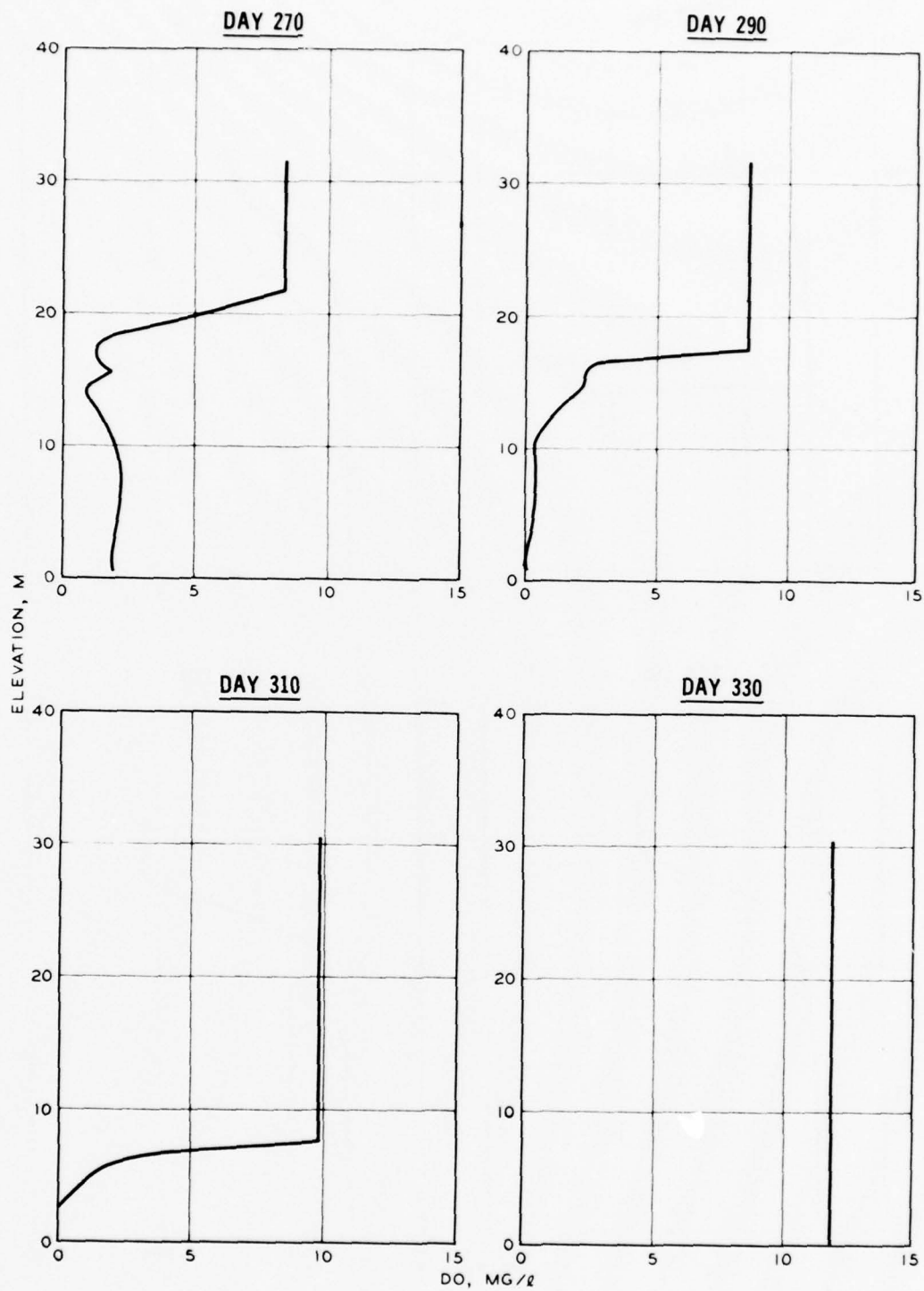


Figure 53 (sheet 3 of 3)

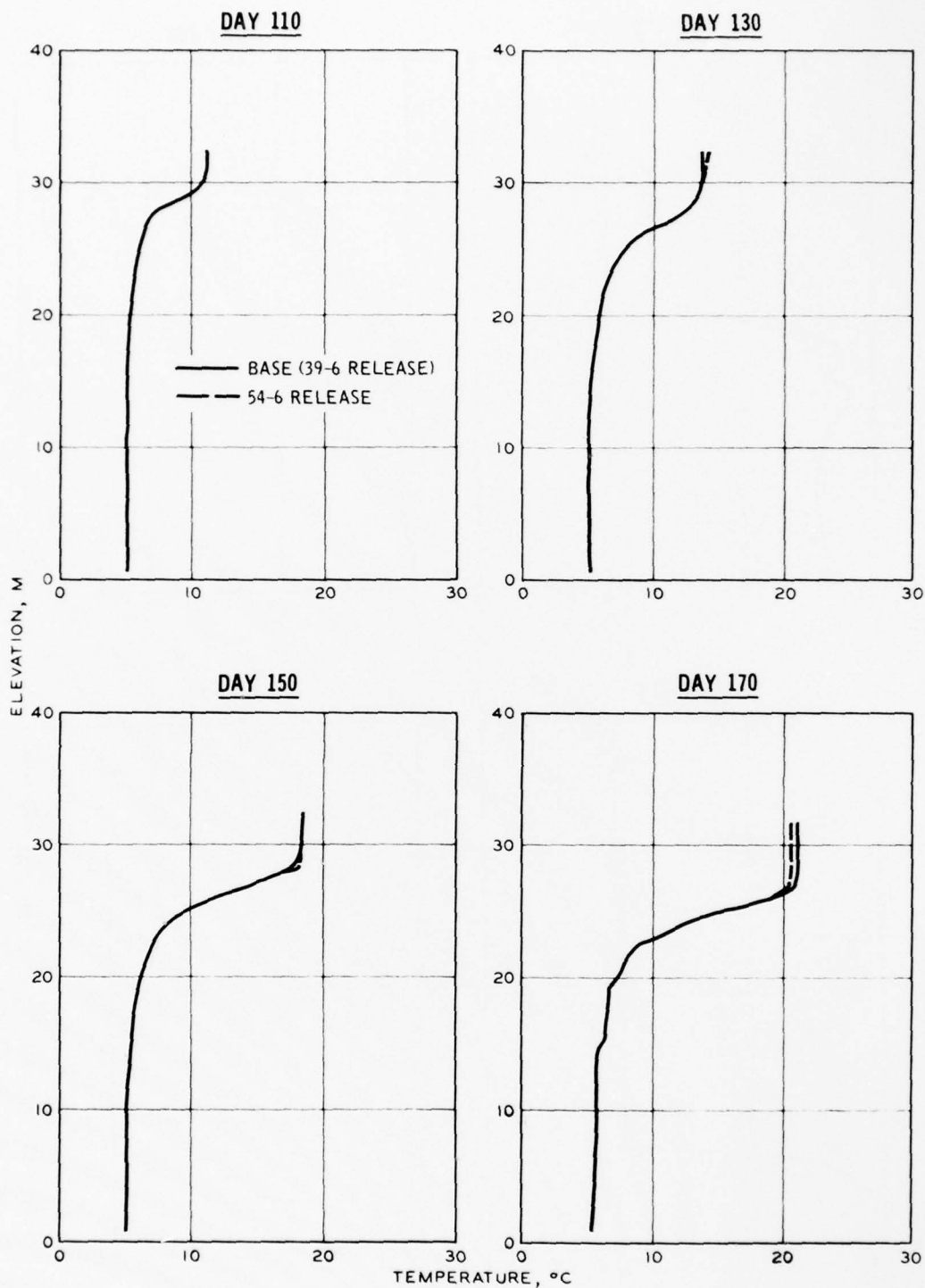


Figure 54. Comparison of 54-6 and base case temperature profiles, 1974 (sheet 1 of 3)

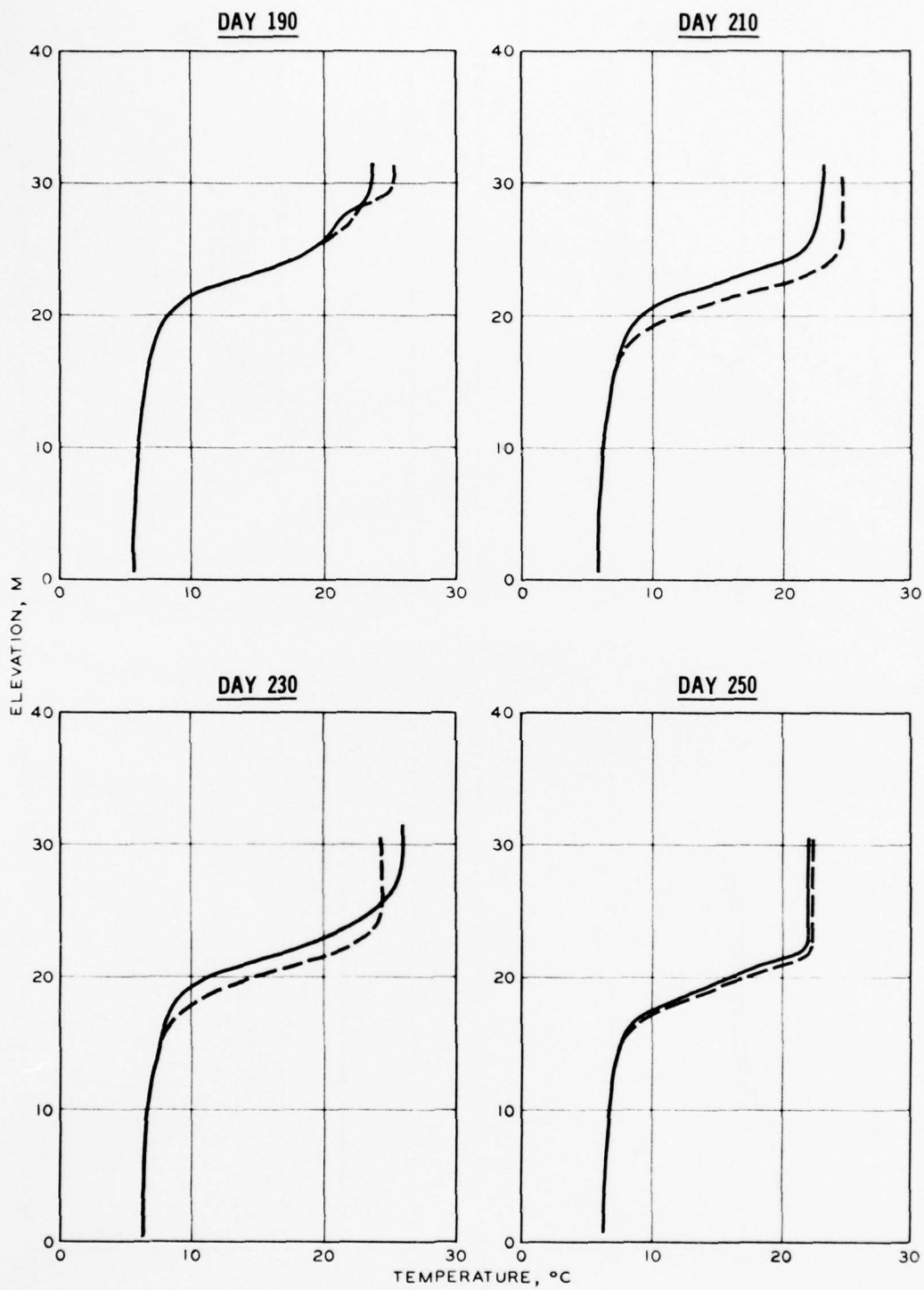


Figure 54 (sheet 2 of 3)

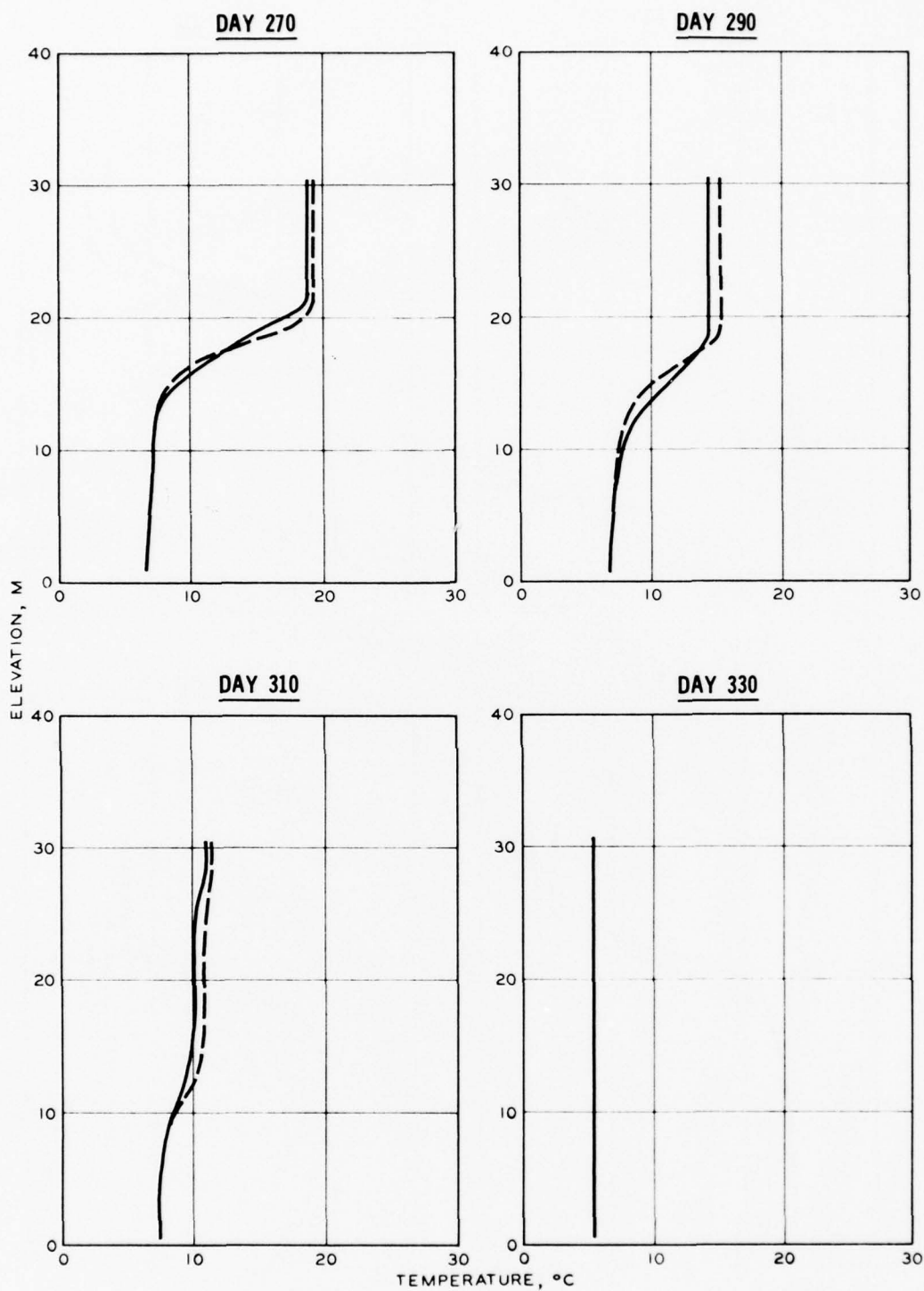


Figure 54 (sheet 3 of 3)

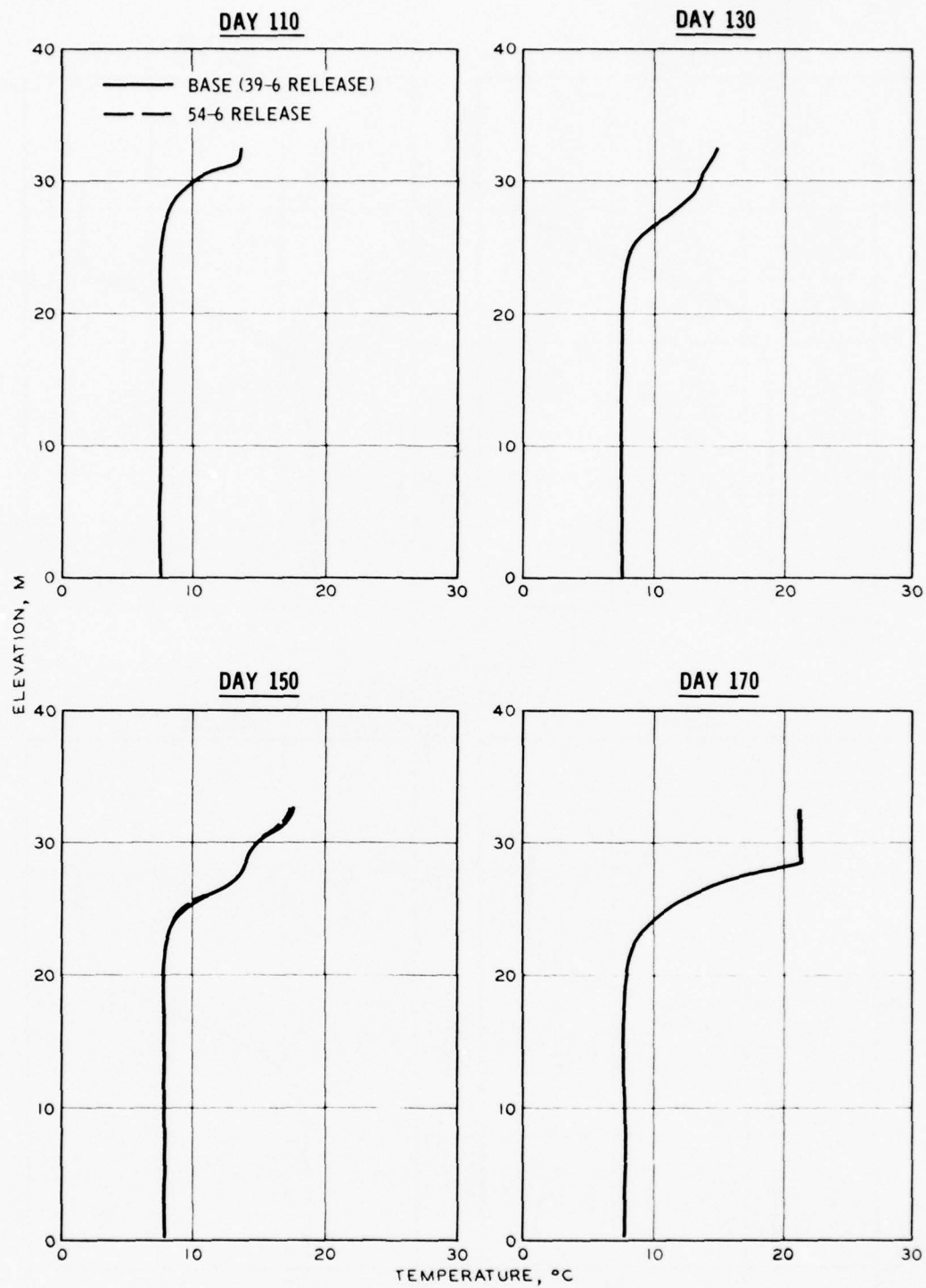


Figure 55. Comparison of 54-6 and base case temperature profiles, 1973 (sheet 1 of 3)

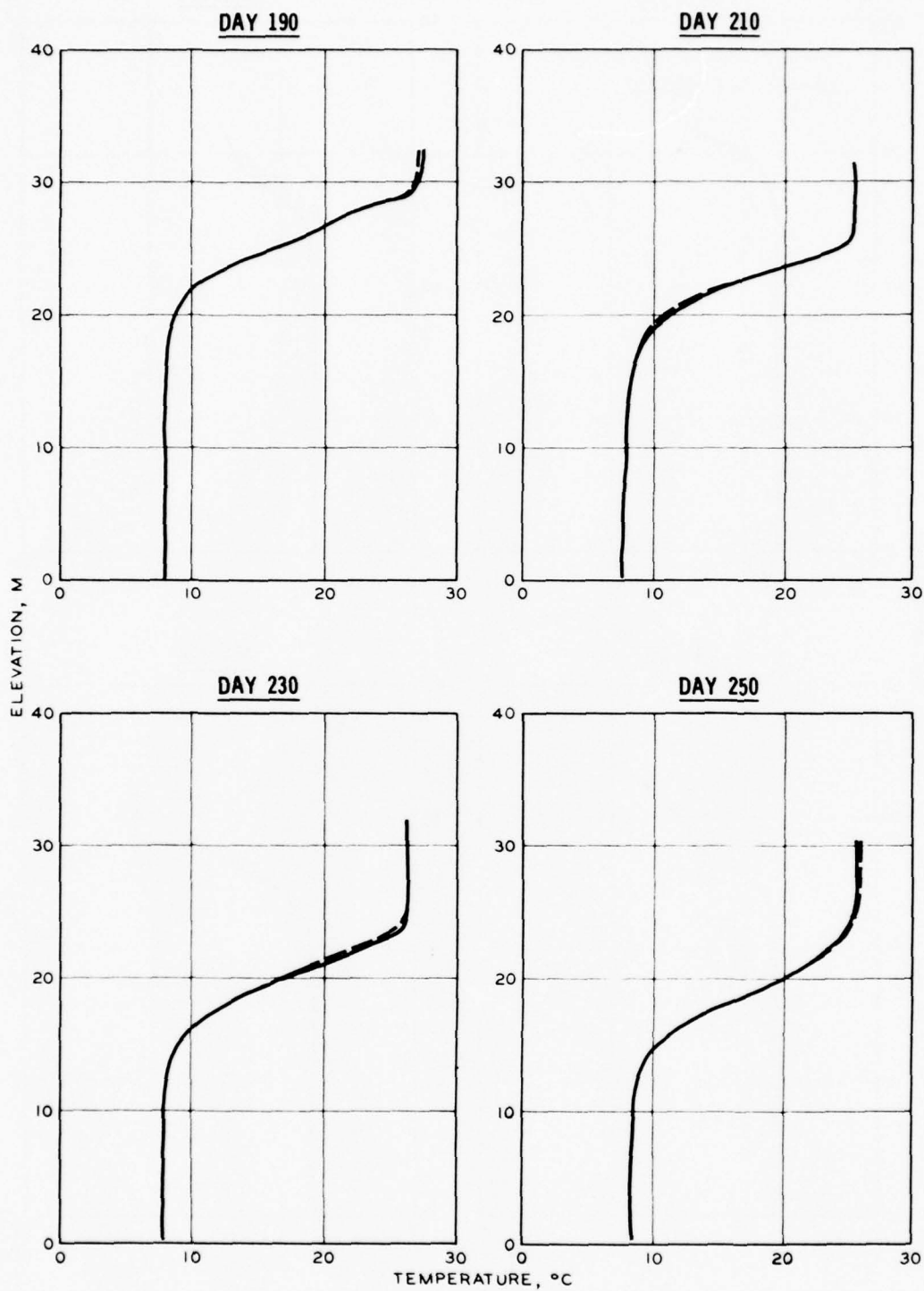


Figure 55 (sheet 2 of 3)

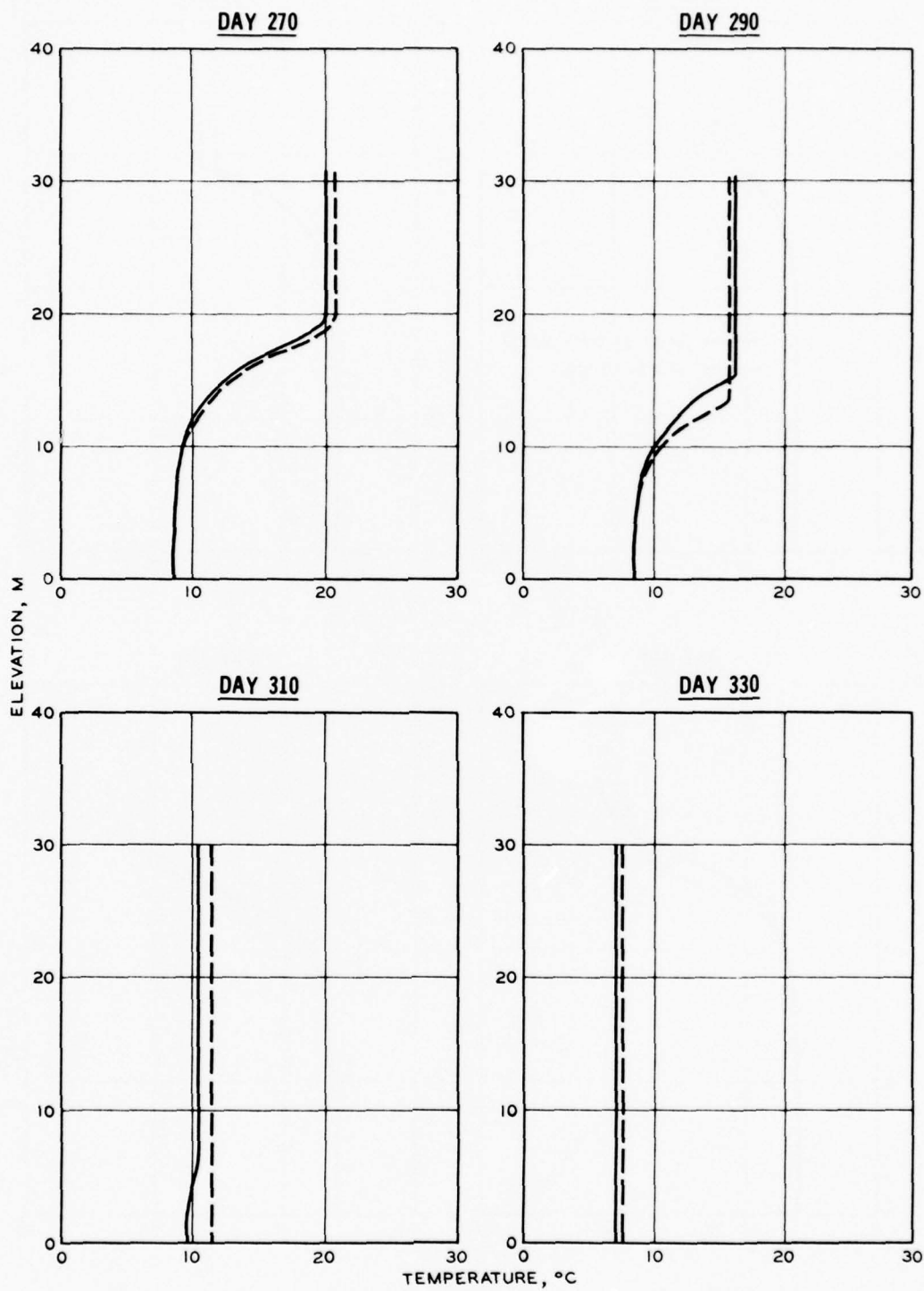


Figure 55 (sheet 3 of 3)

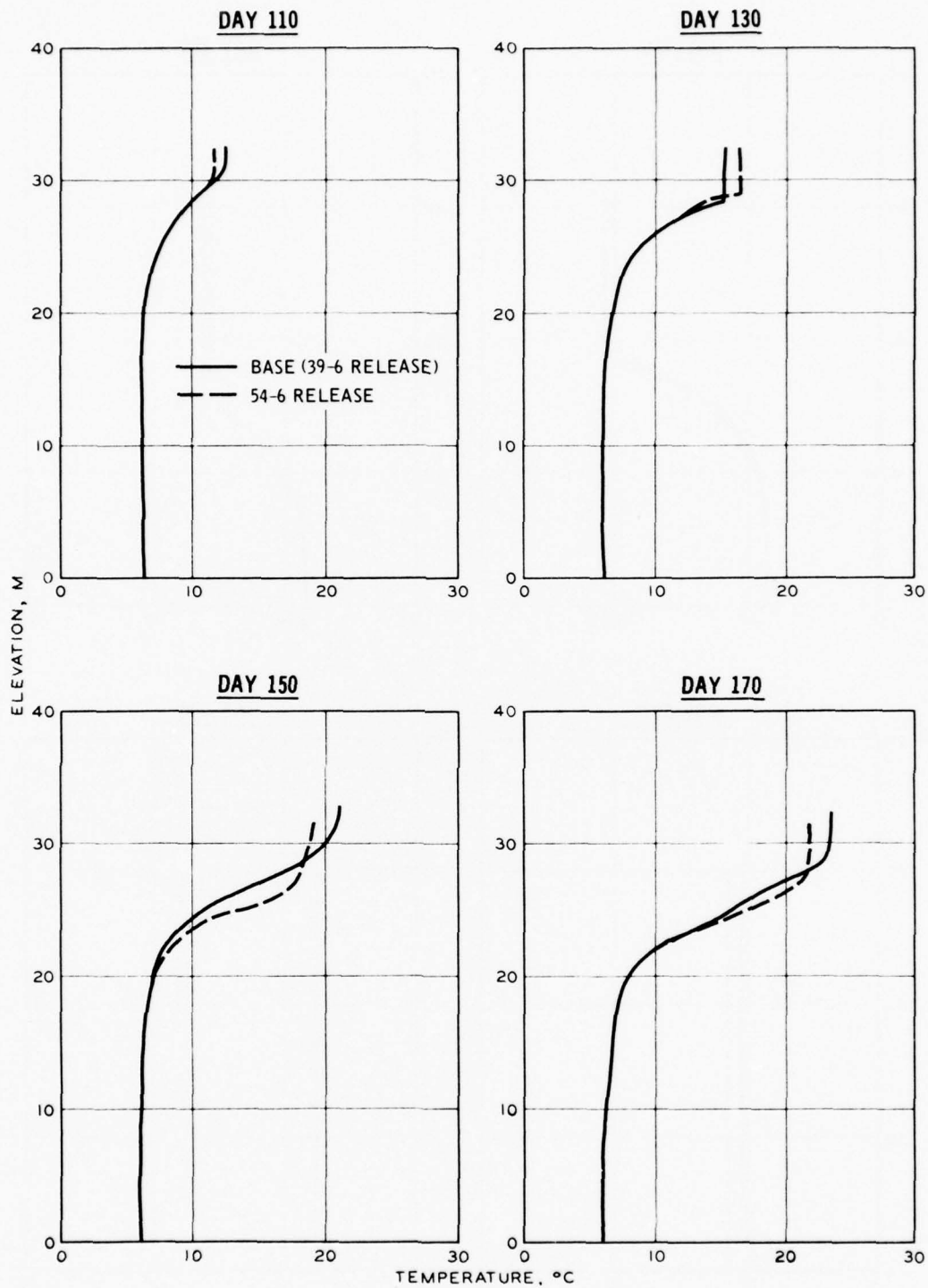


Figure 56. Comparison of 54-6 and base case temperature profiles, 1969 (sheet 1 of 3)

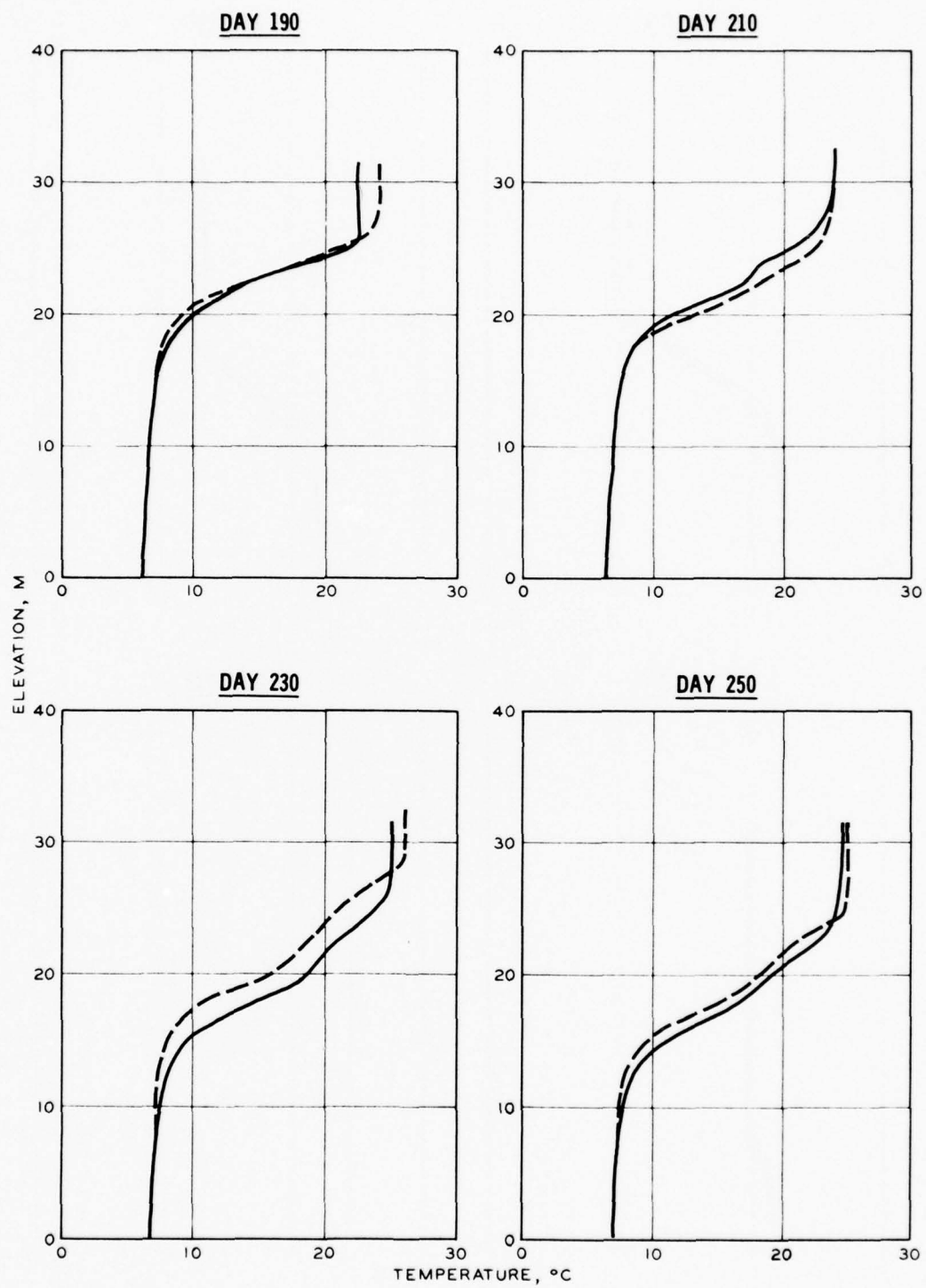


Figure 56 (sheet 2 of 3)

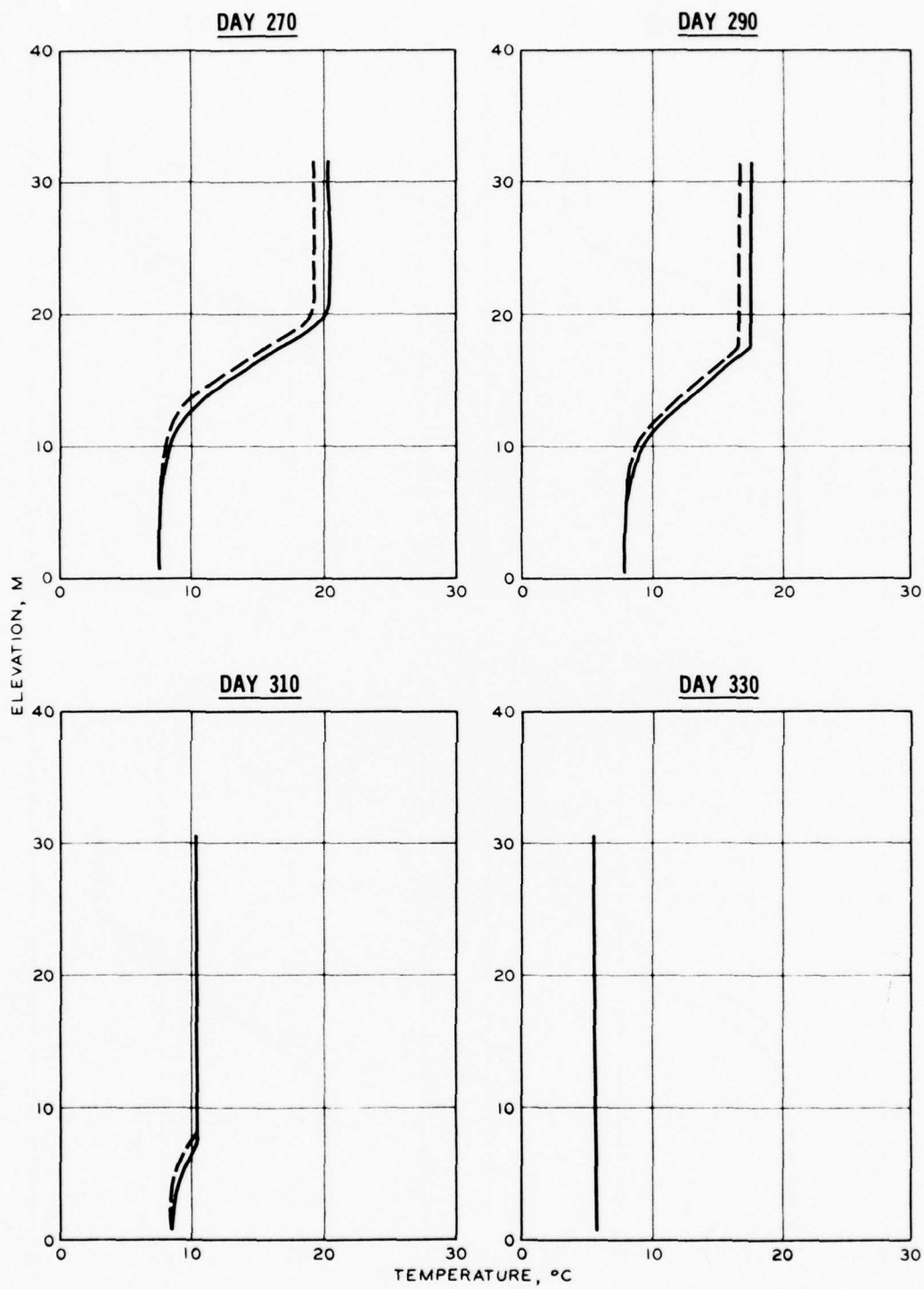


Figure 56 (sheet 3 of 3)

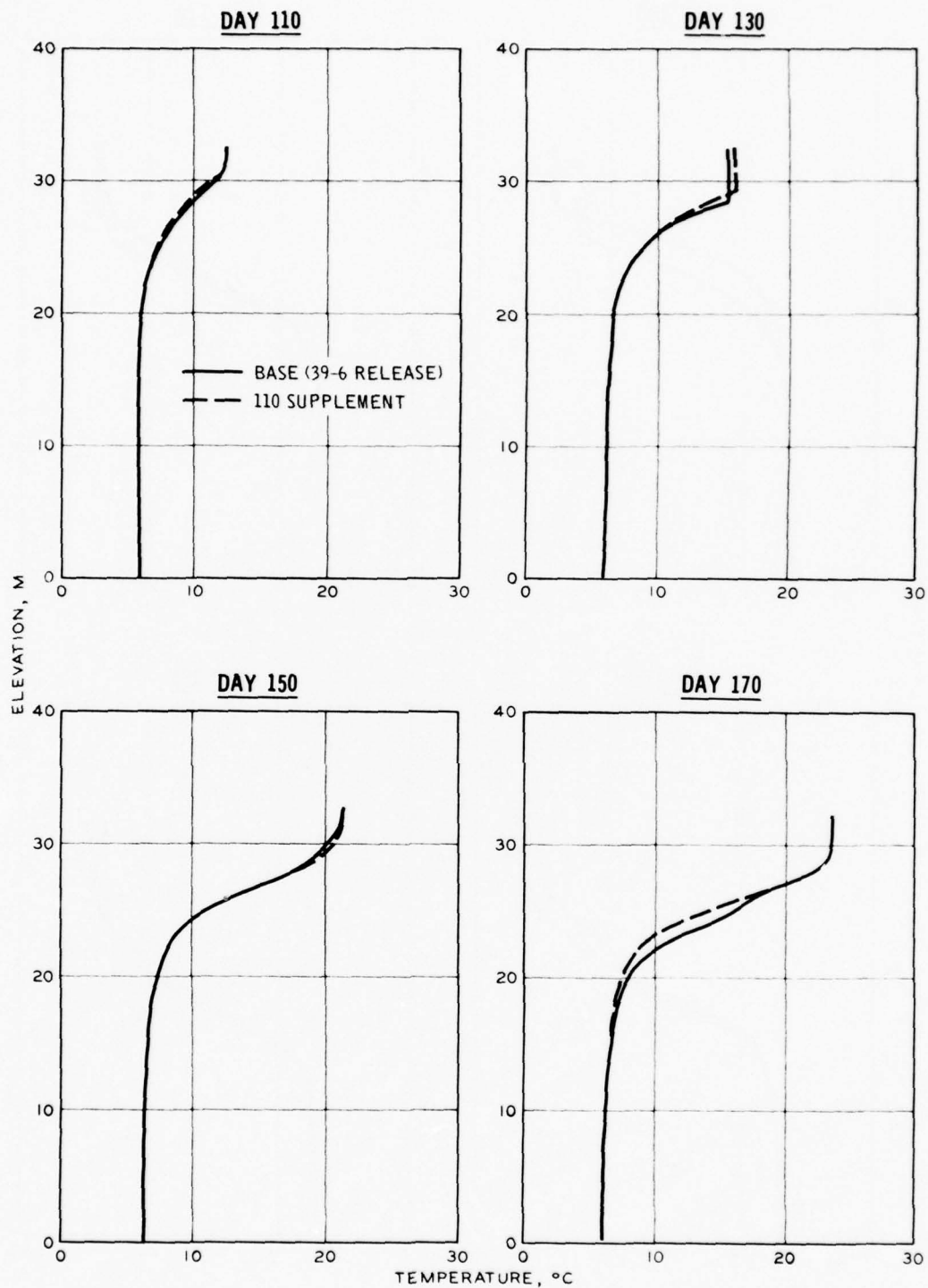


Figure 57. Comparison of 110 Trenton supplement and base case temperature profiles, 1969 (sheet 1 of 3)

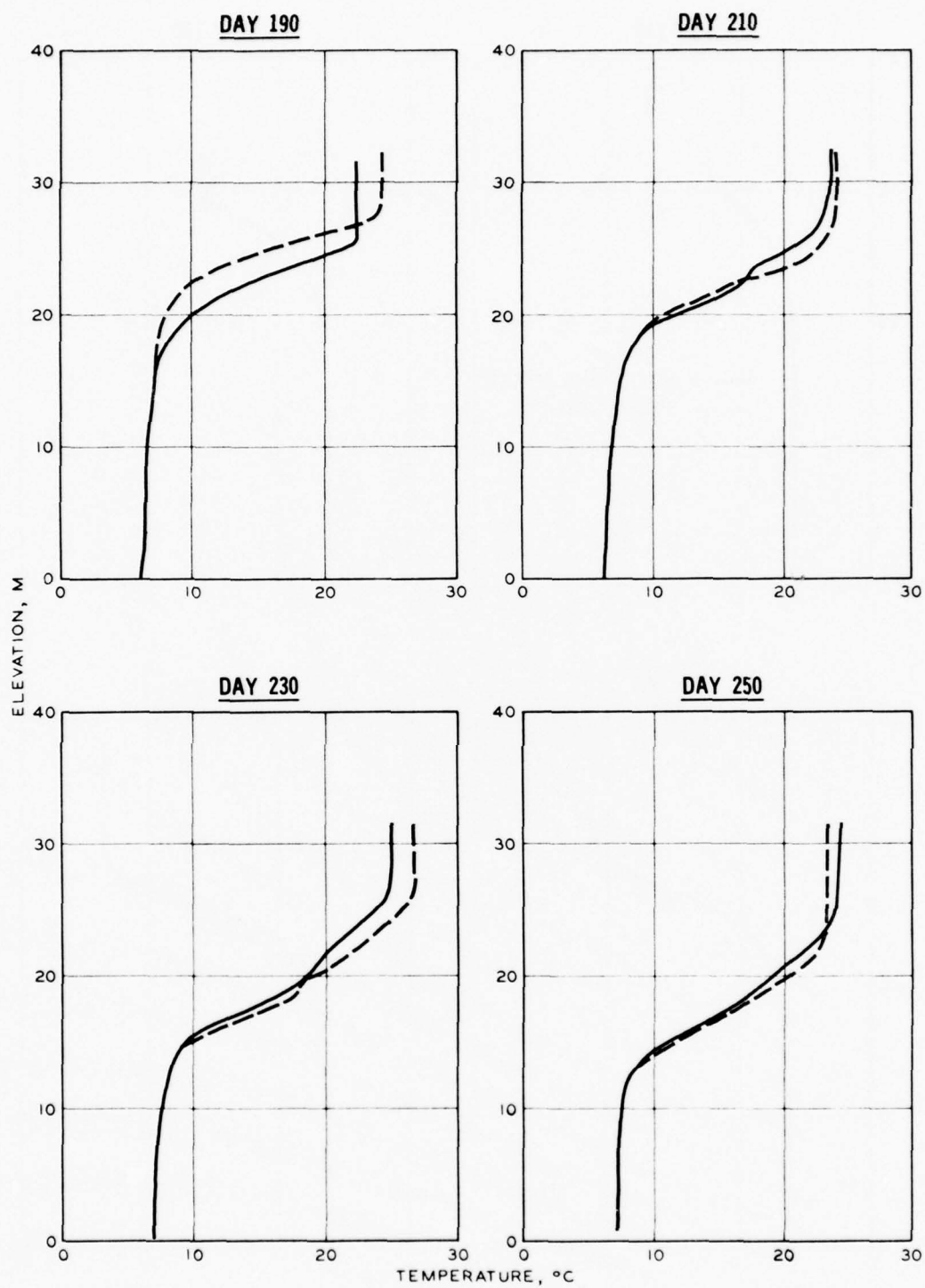


Figure 57 (sheet 2 of 3)

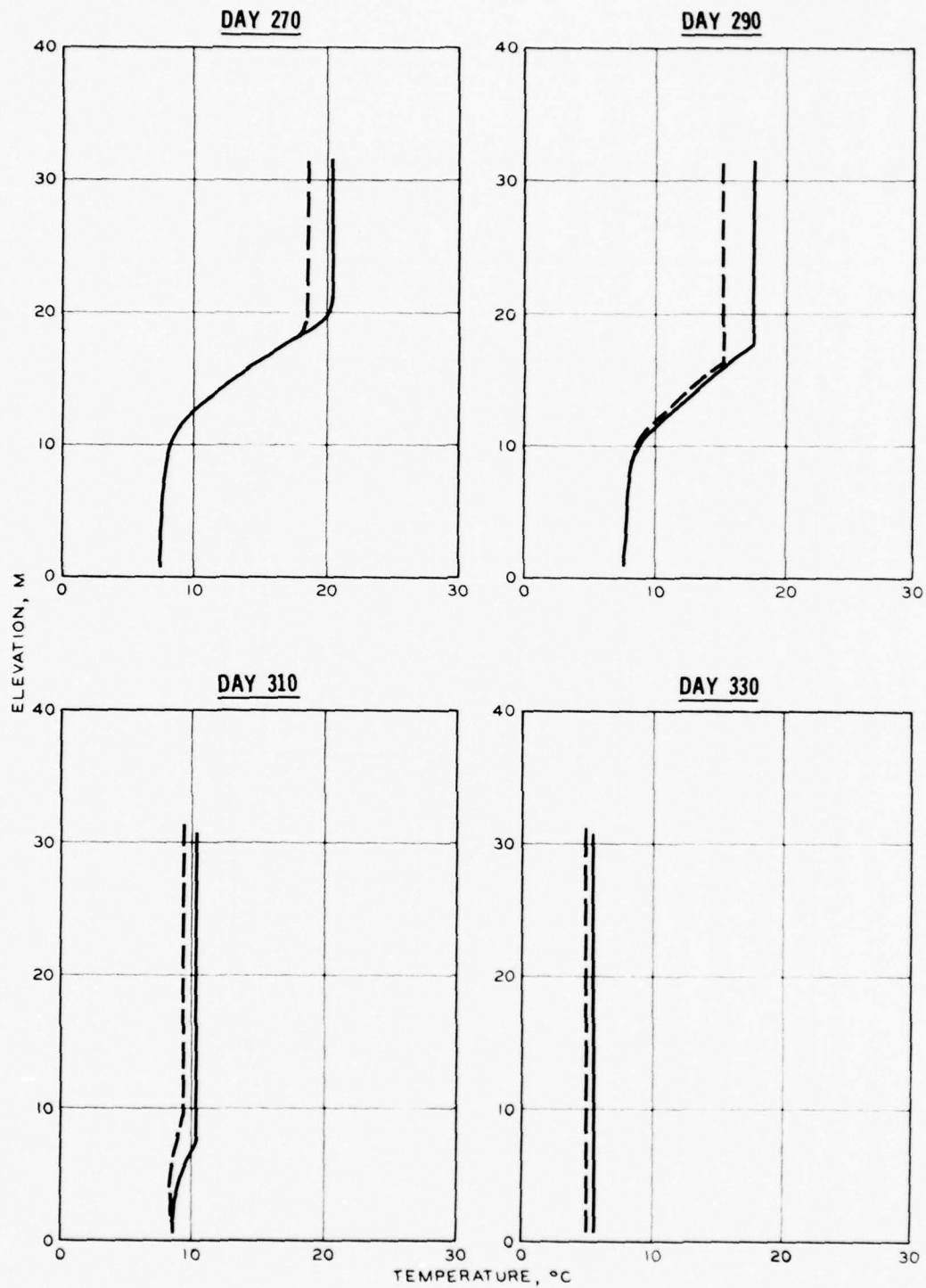


Figure 57 (sheet 3 of 3)

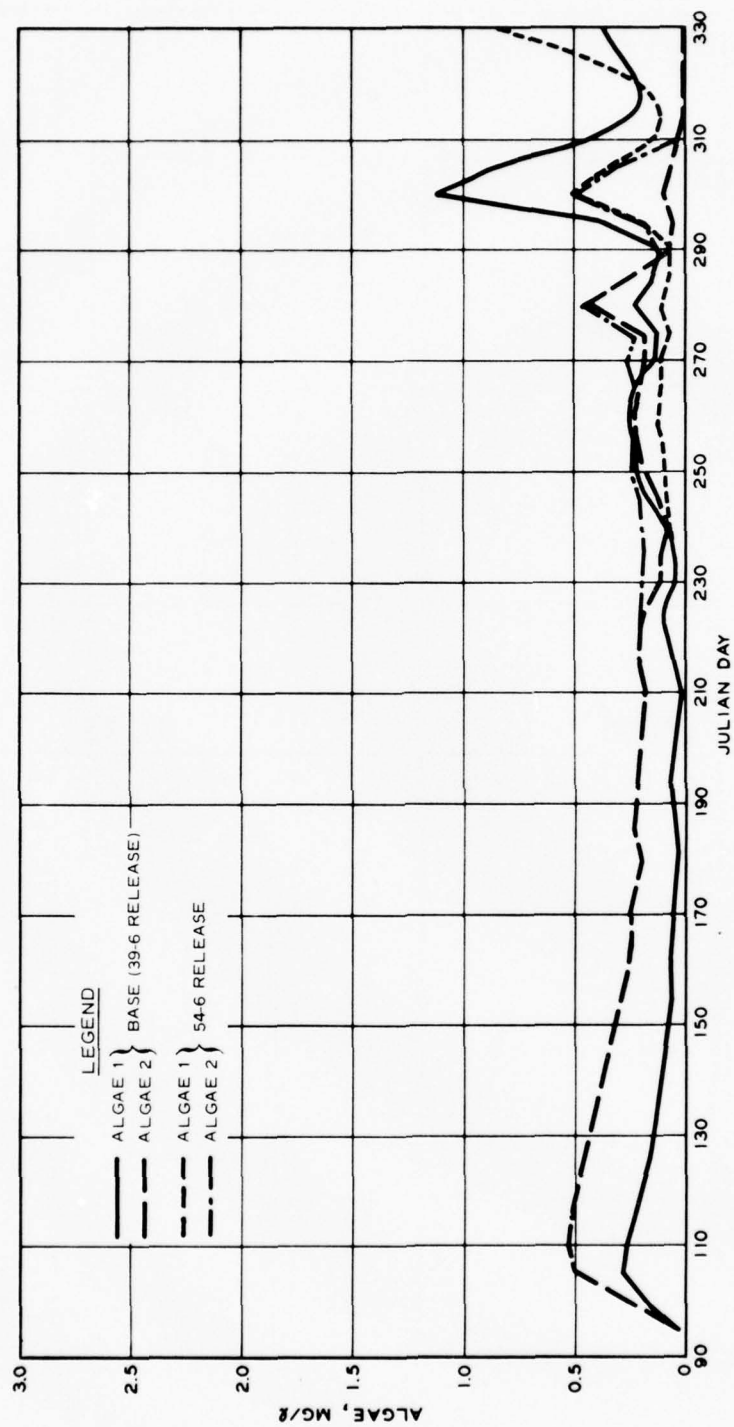


Figure 58. Comparison of algae concentrations for the 54-6 release schedule and base case, 1974

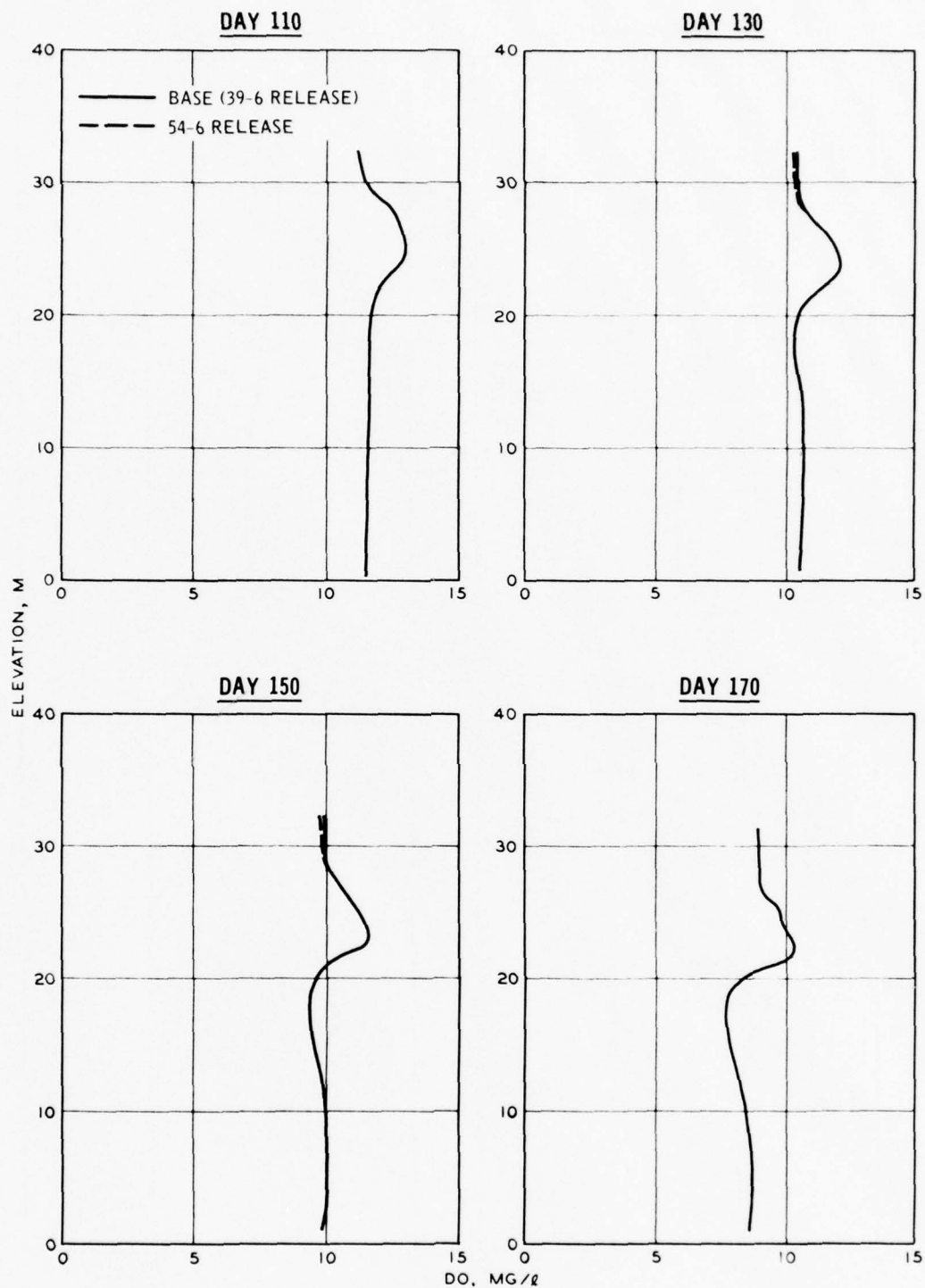


Figure 59. Comparison of DO profiles for the 54-6 release schedule and base case, 1974 (sheet 1 of 3)

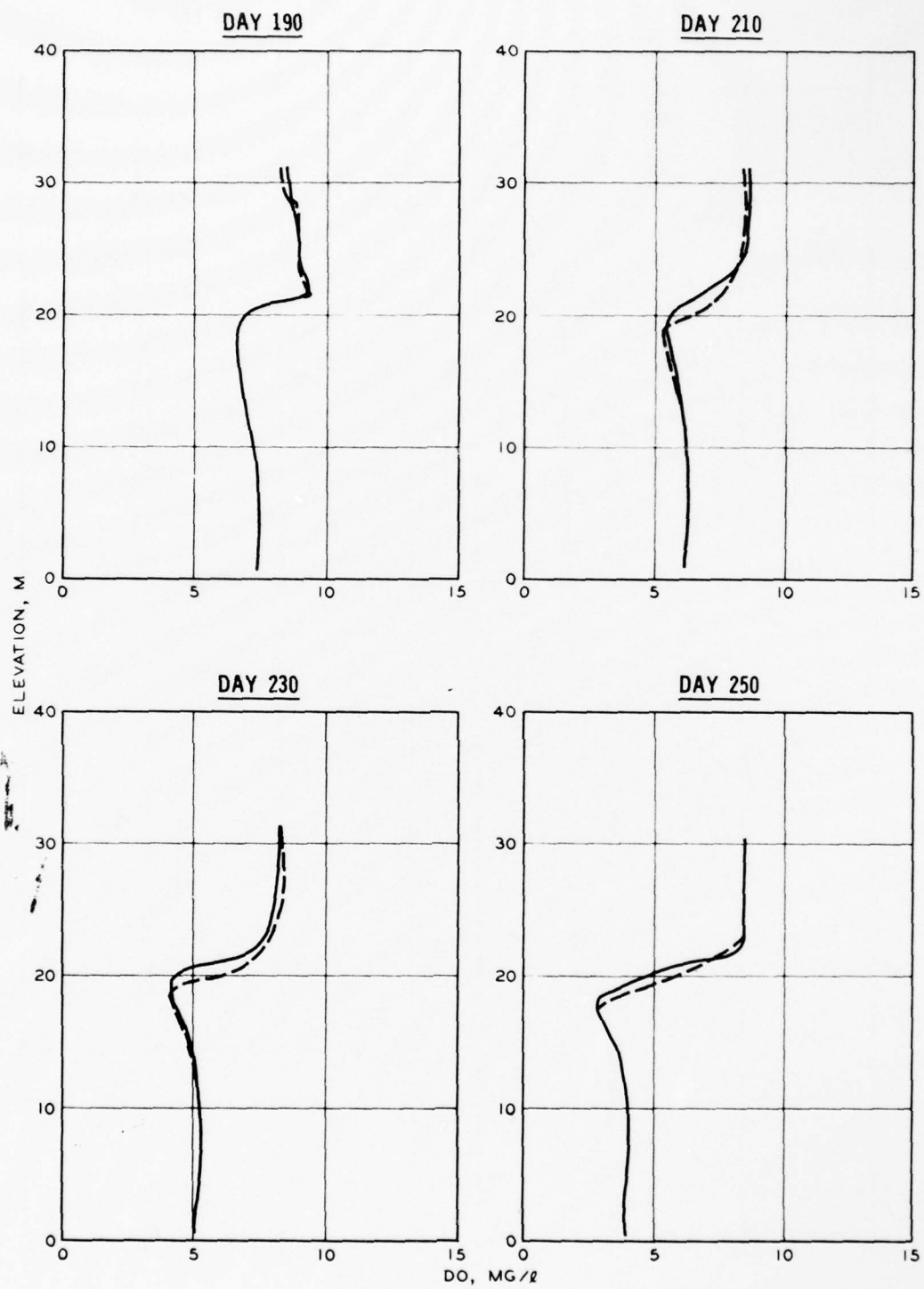


Figure 59 (sheet 2 of 3)

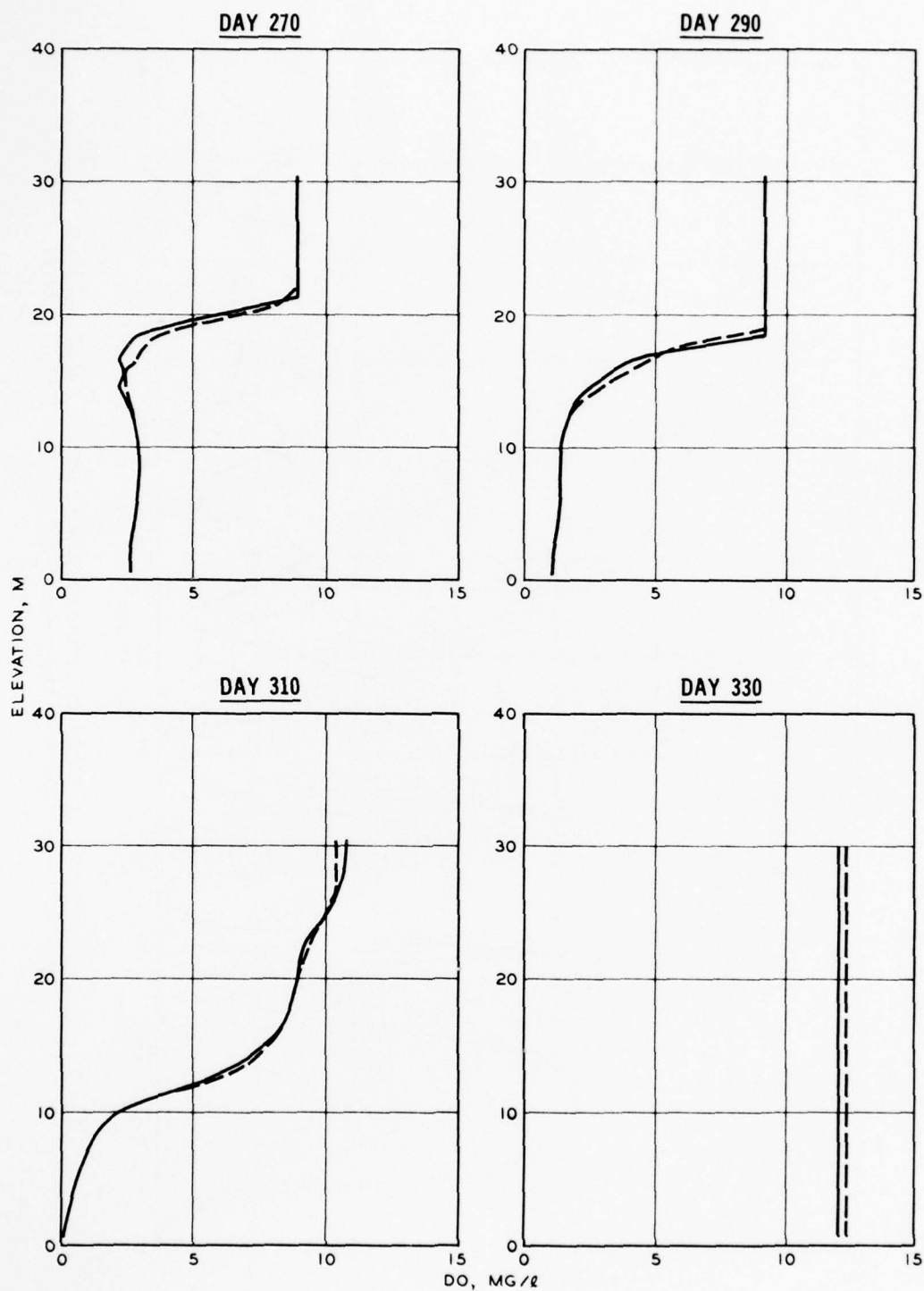


Figure 59 (sheet 3 of 3)

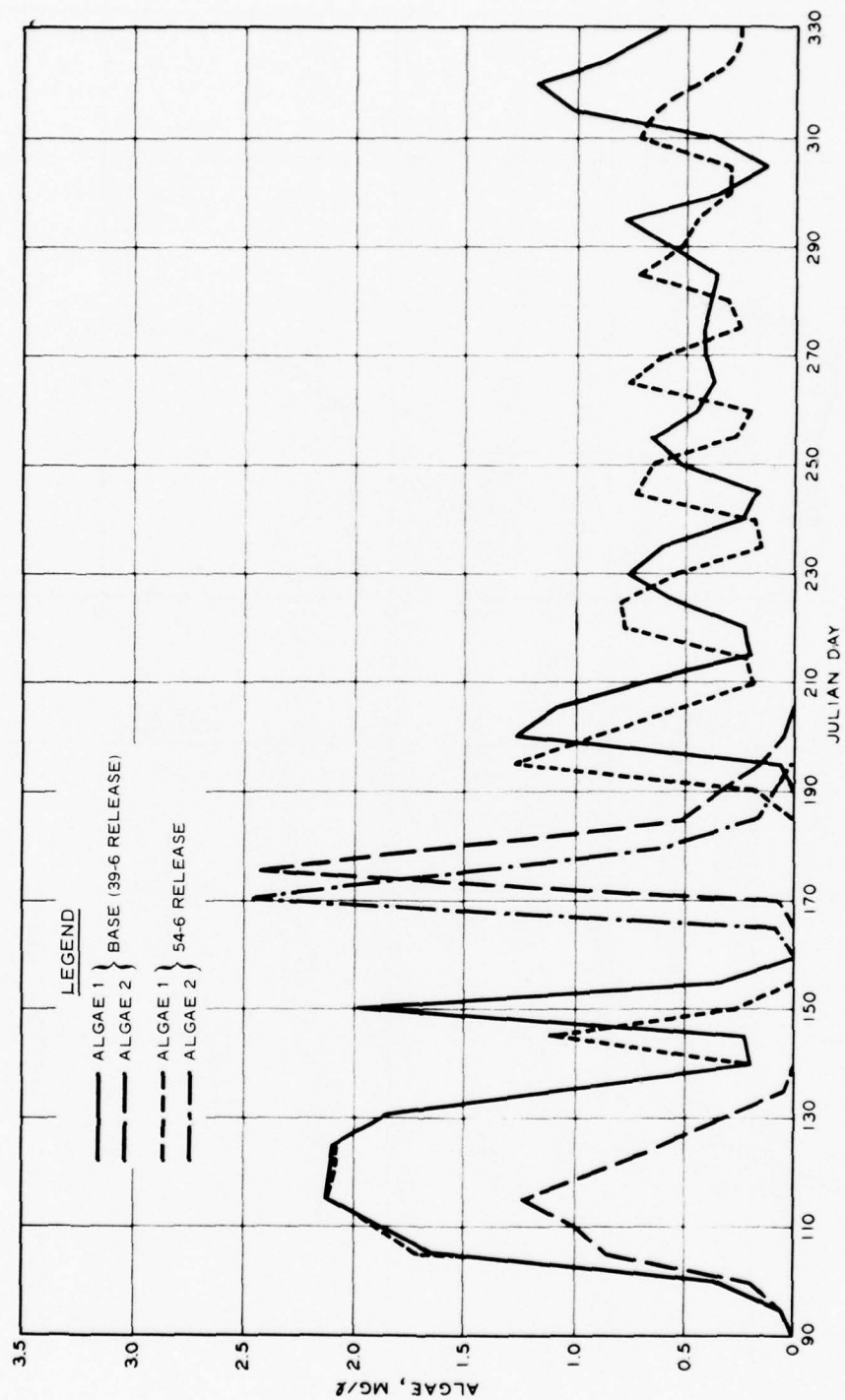


Figure 60. Comparison of algae concentrations under the 54-6 release schedule and base case, 1973

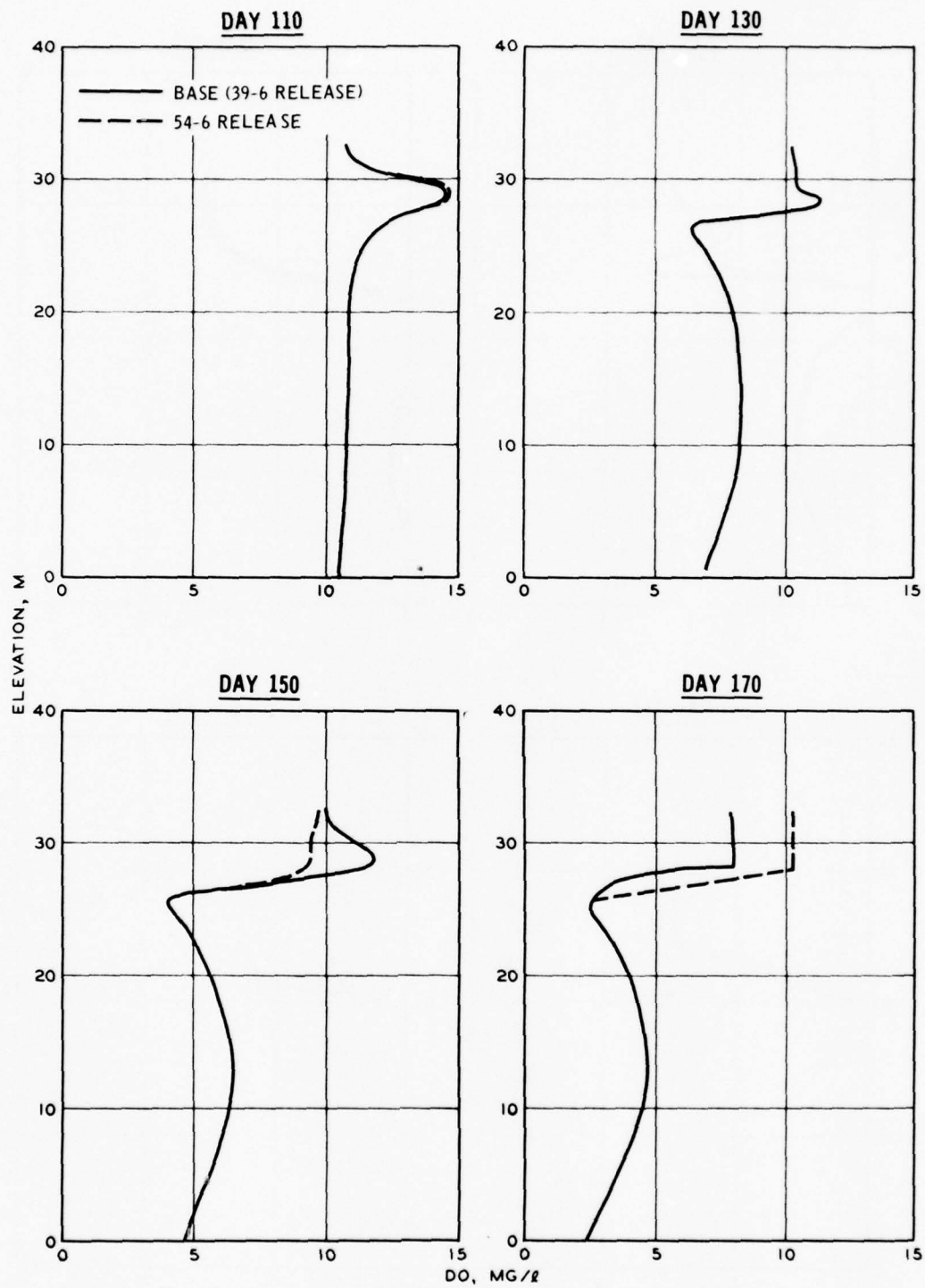


Figure 61. Comparison of DO profiles under the 54-6 release schedule and base case, 1973 (sheet 1 of 3)

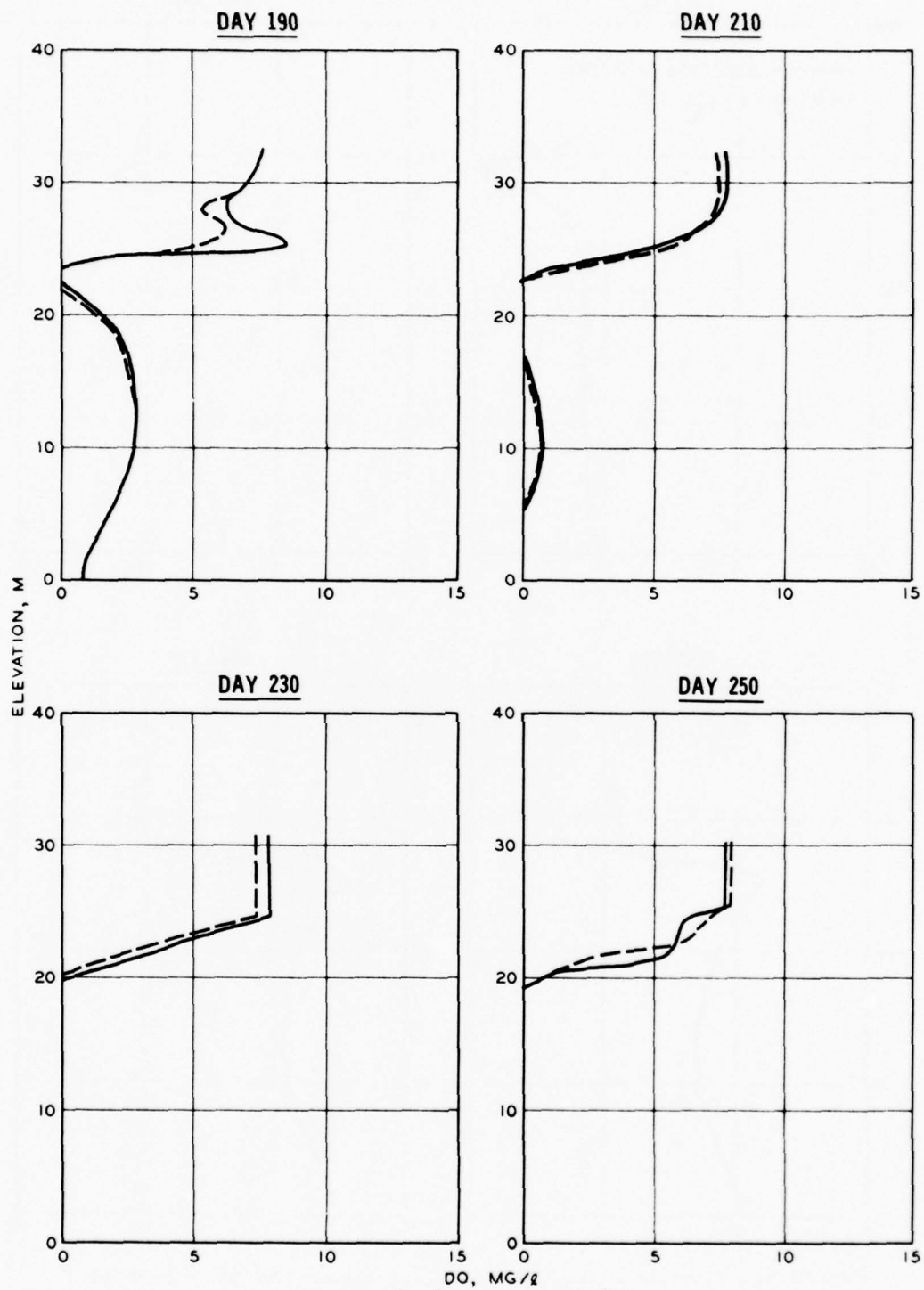


Figure 61 (sheet 2 of 3)

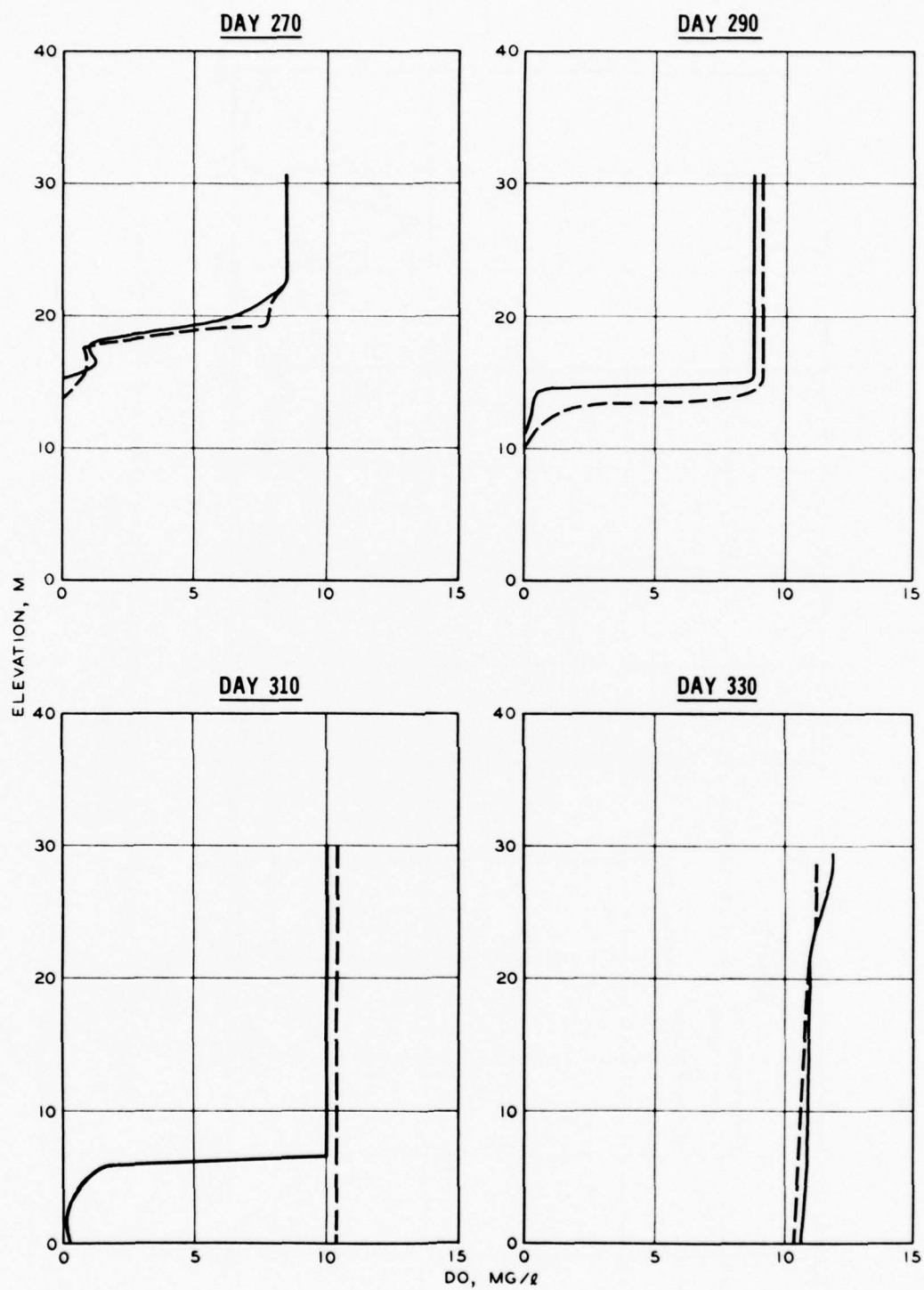


Figure 61 (sheet 3 of 3)

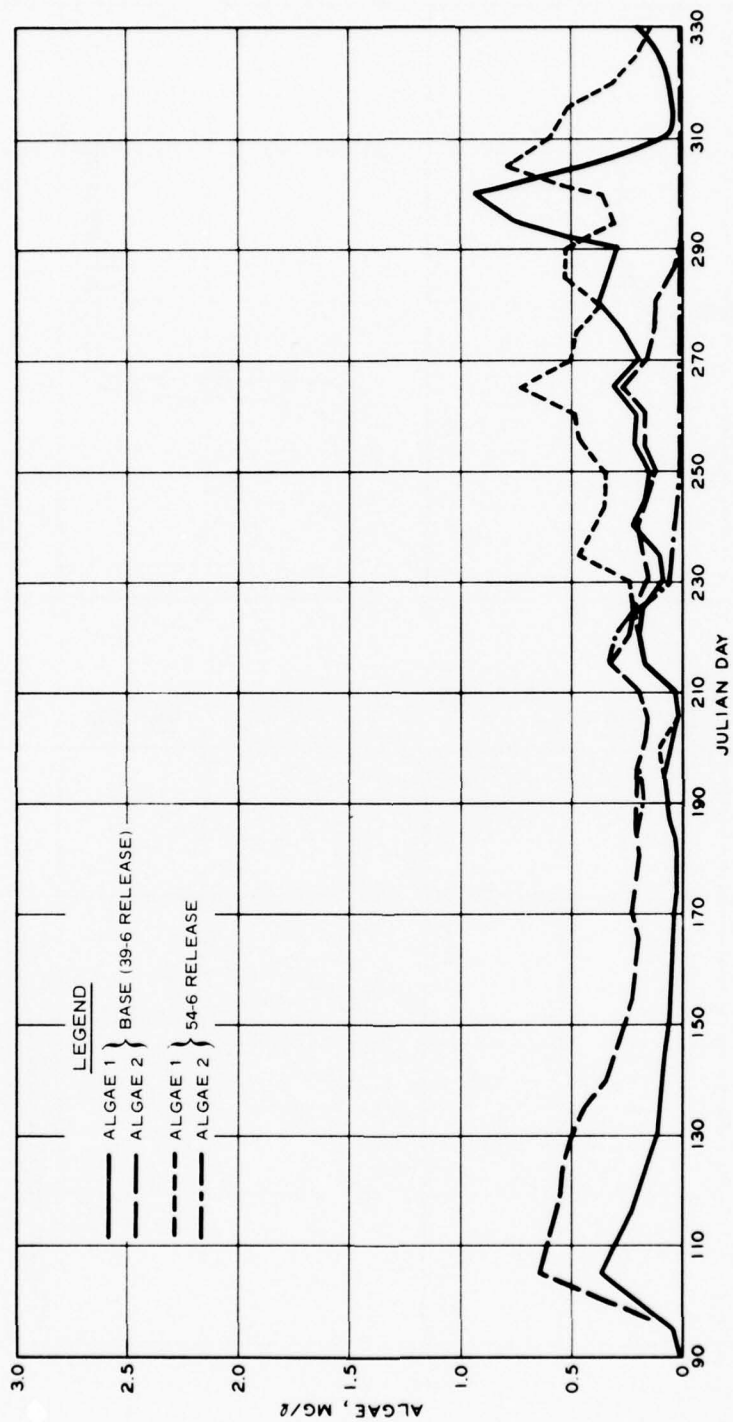


Figure 62. Comparison of algae concentrations under the 54-6 release schedule and base case, 1969

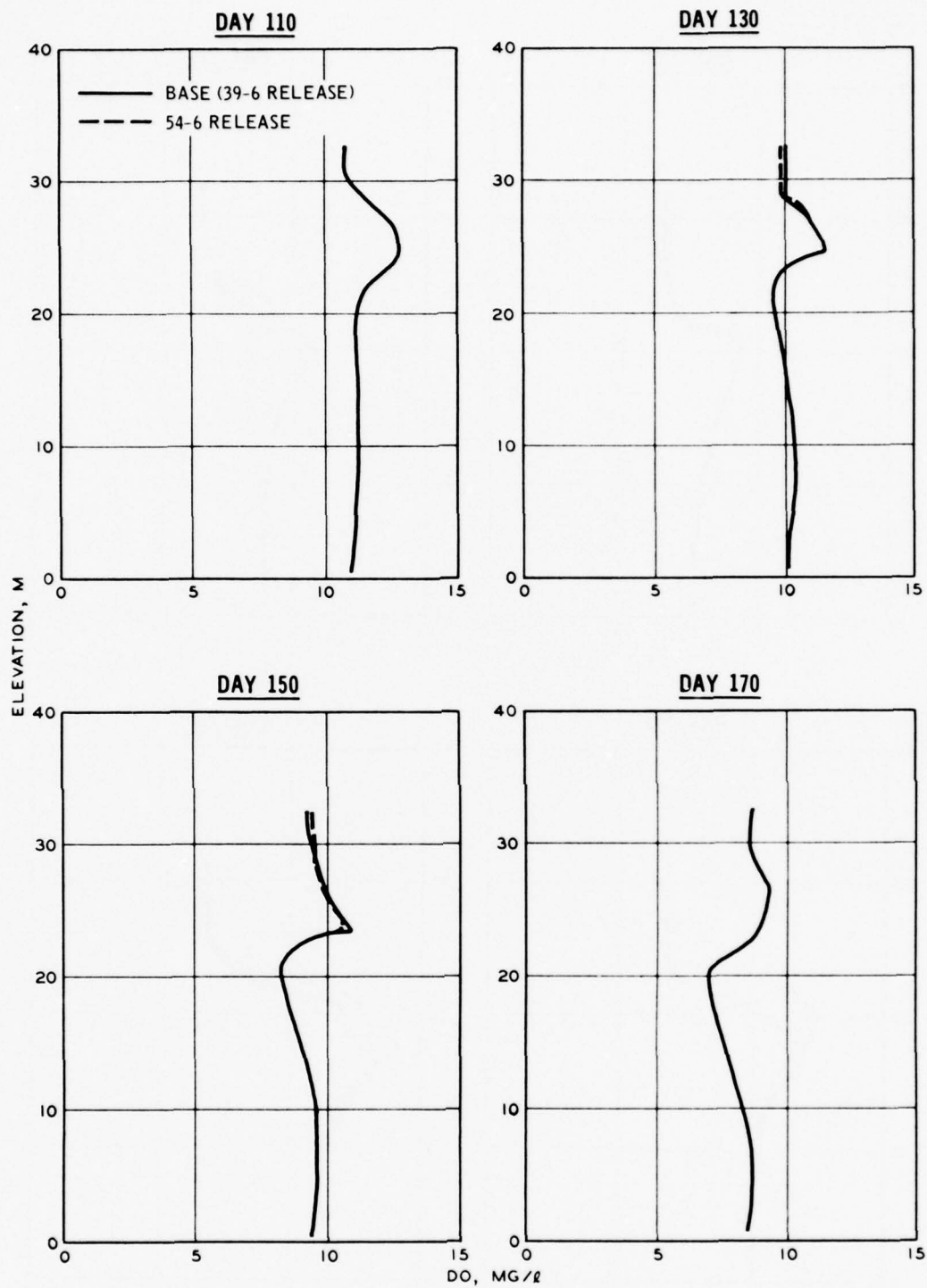


Figure 63. Comparison of DO profiles under the 54-6 release schedule and base case, 1969 (sheet 1 of 3)

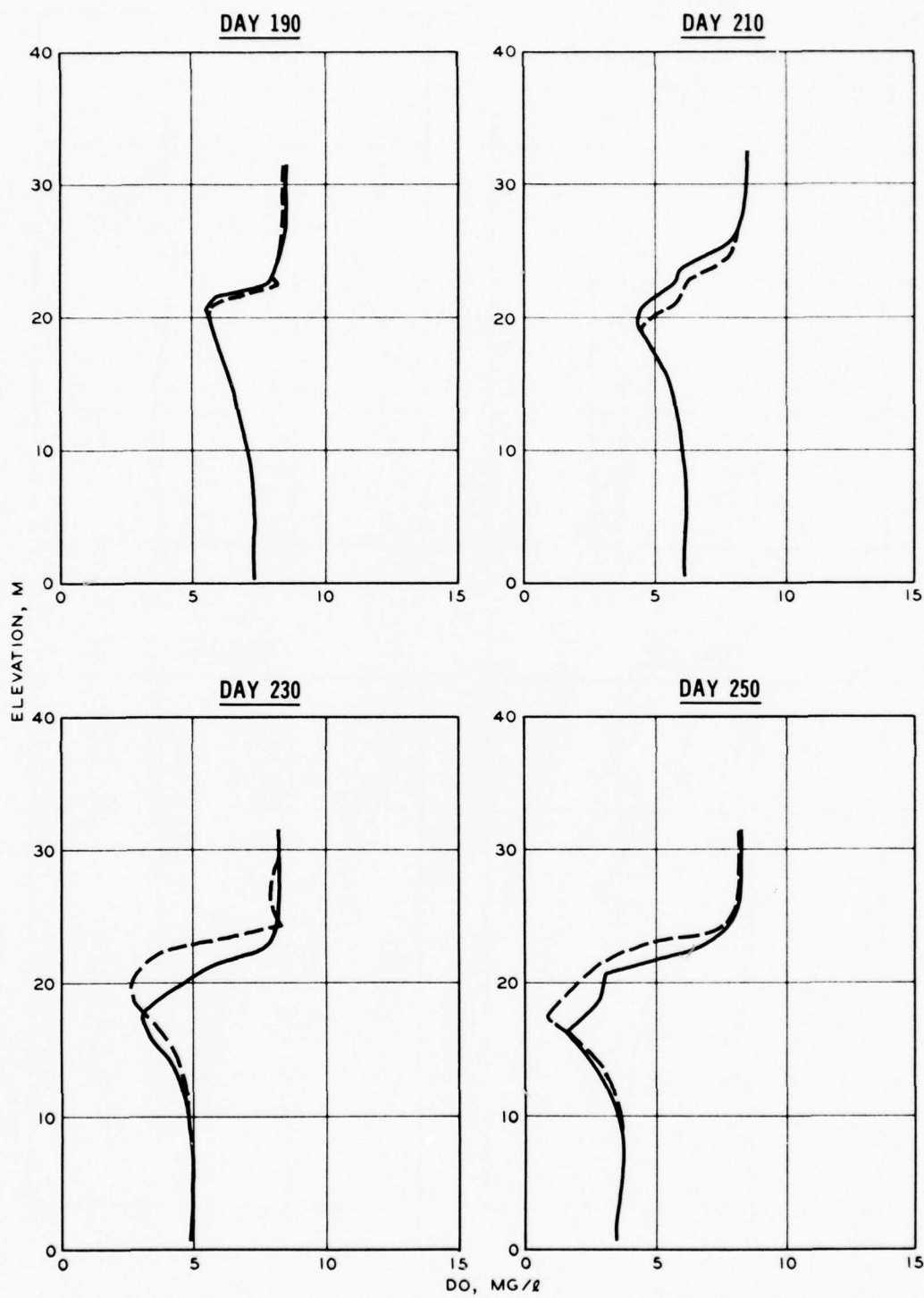


Figure 63 (sheet 2 of 3)

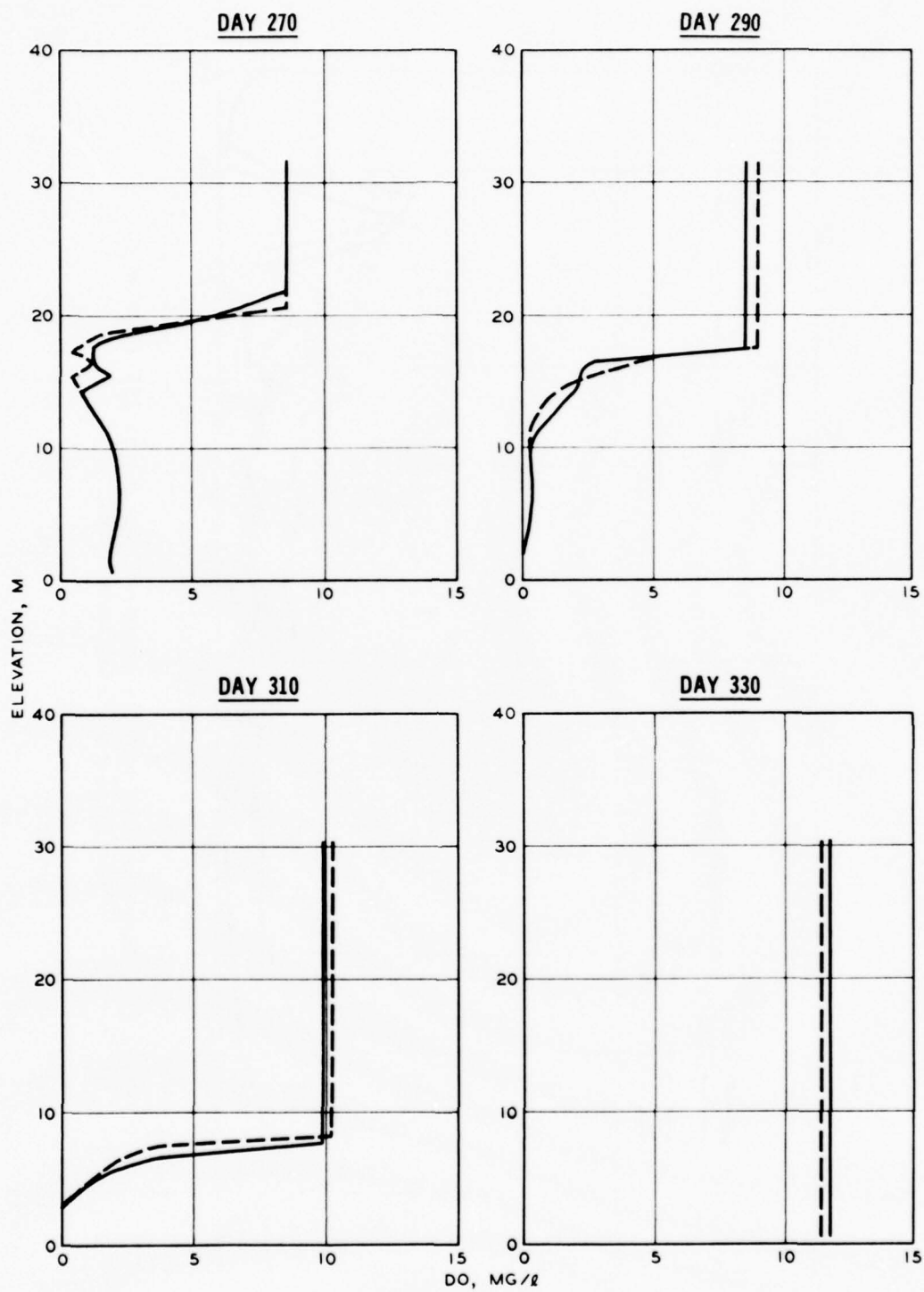


Figure 63 (sheet 3 of 3)

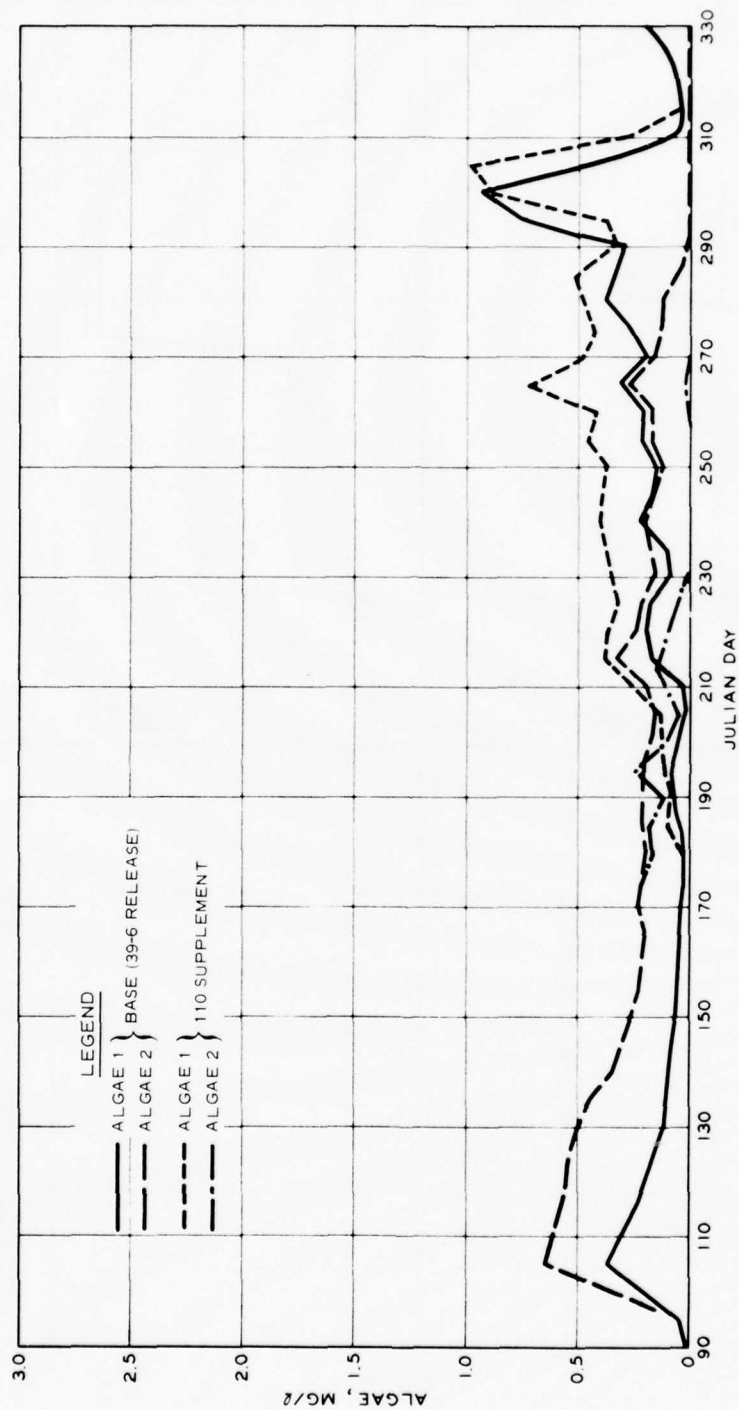


Figure 64. Comparison of algae concentrations in the 110 Trenton supplement and base case, 1969

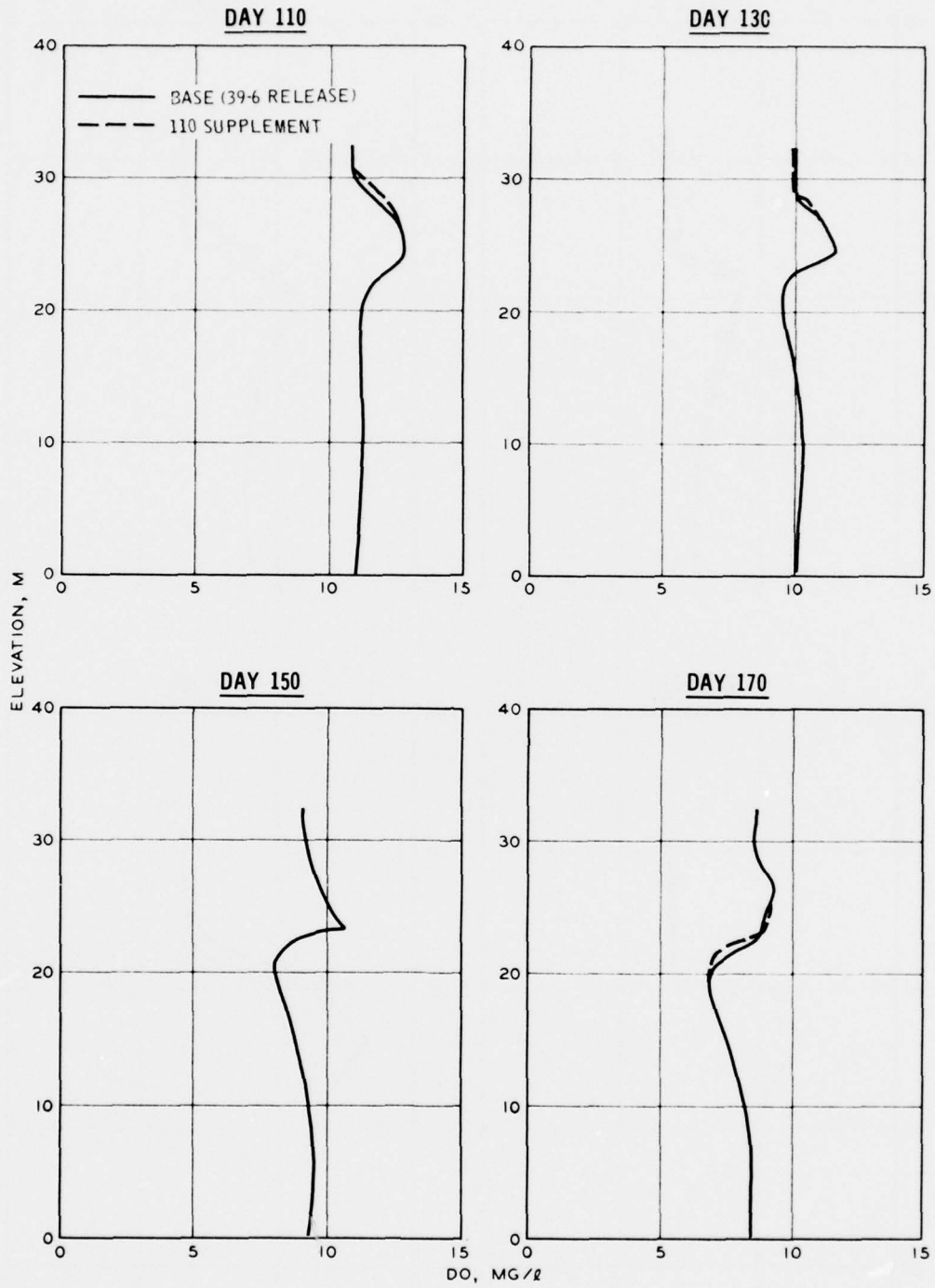


Figure 65. Comparison of DO profiles of the 110 Trenton supplement and base case, 1969 (sheet 1 of 3)

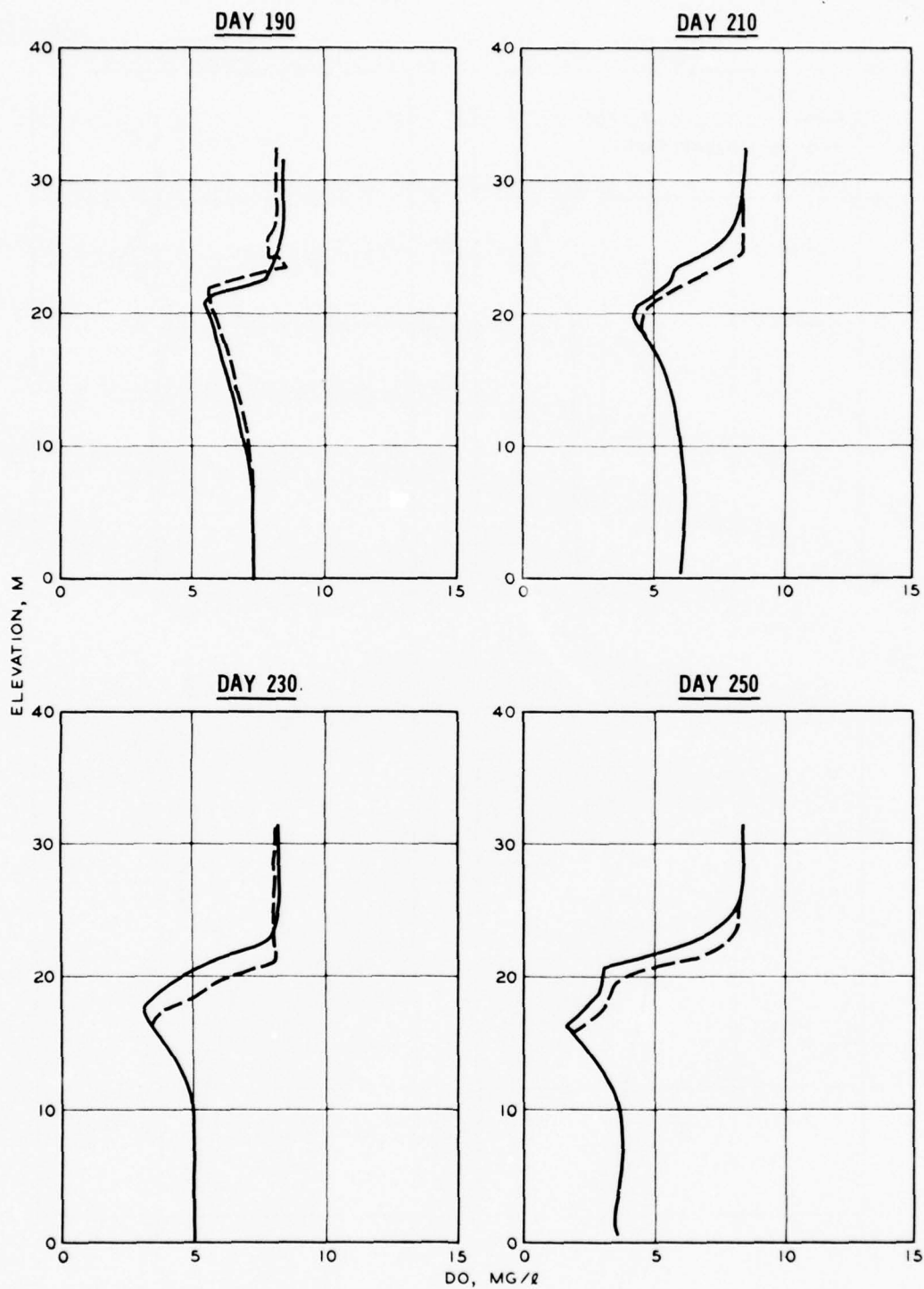


Figure 65 (sheet 2 of 3)

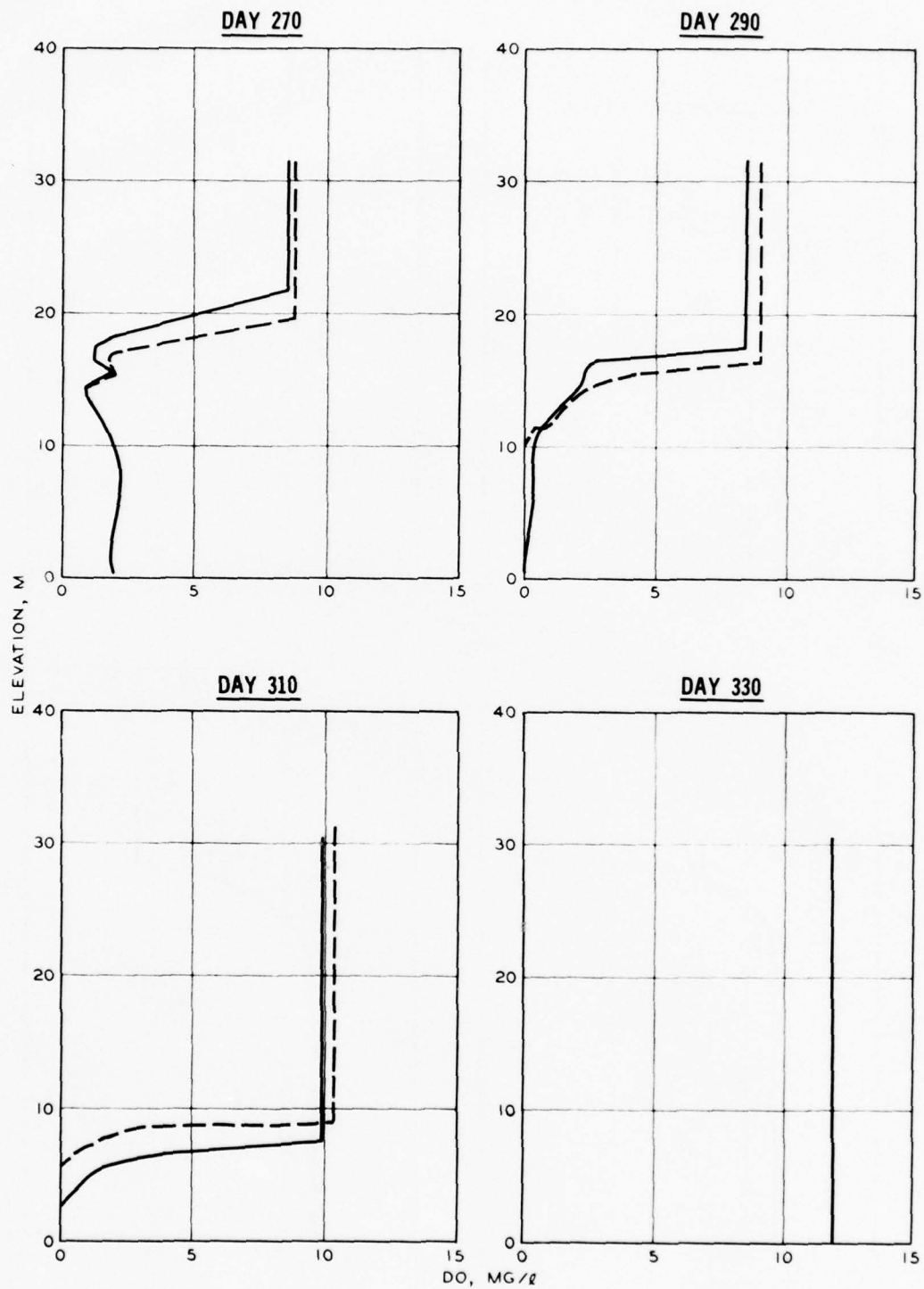


Figure 65 (sheet 3 of 3)

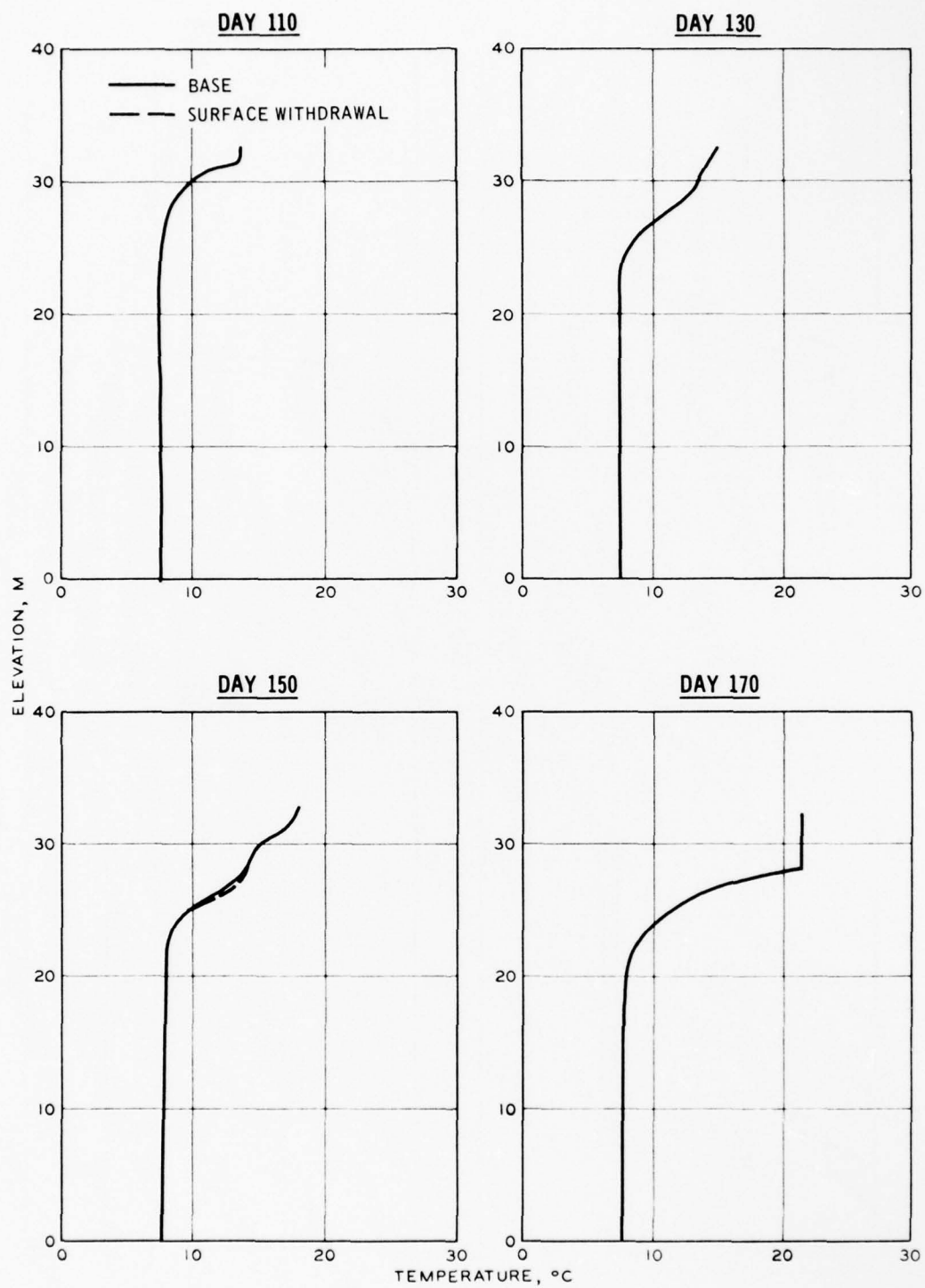


Figure 66. Comparison of temperature profiles using surface withdrawal and selective withdrawal (base) (sheet 1 of 3)

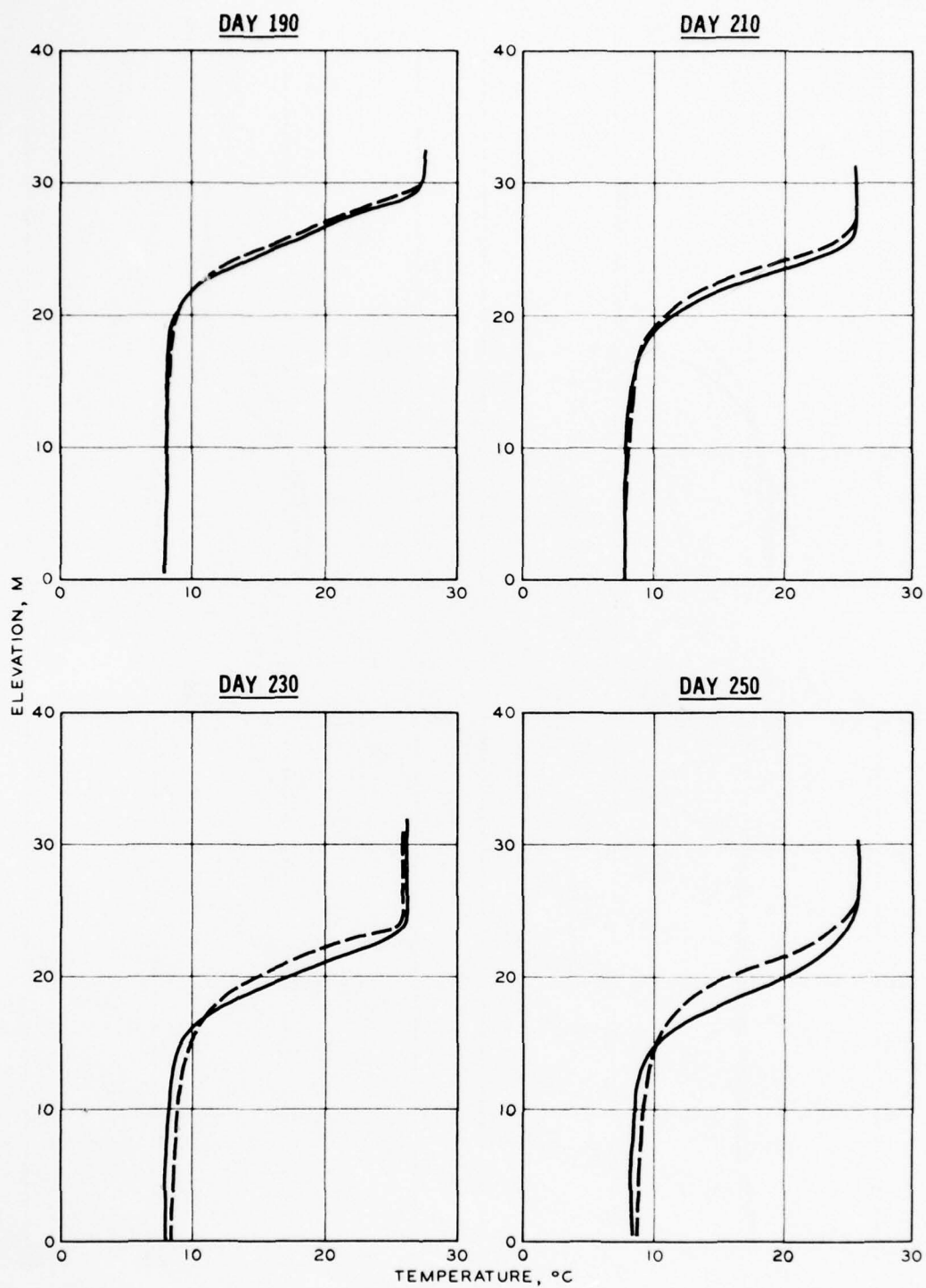


Figure 66 (sheet 2 of 3)

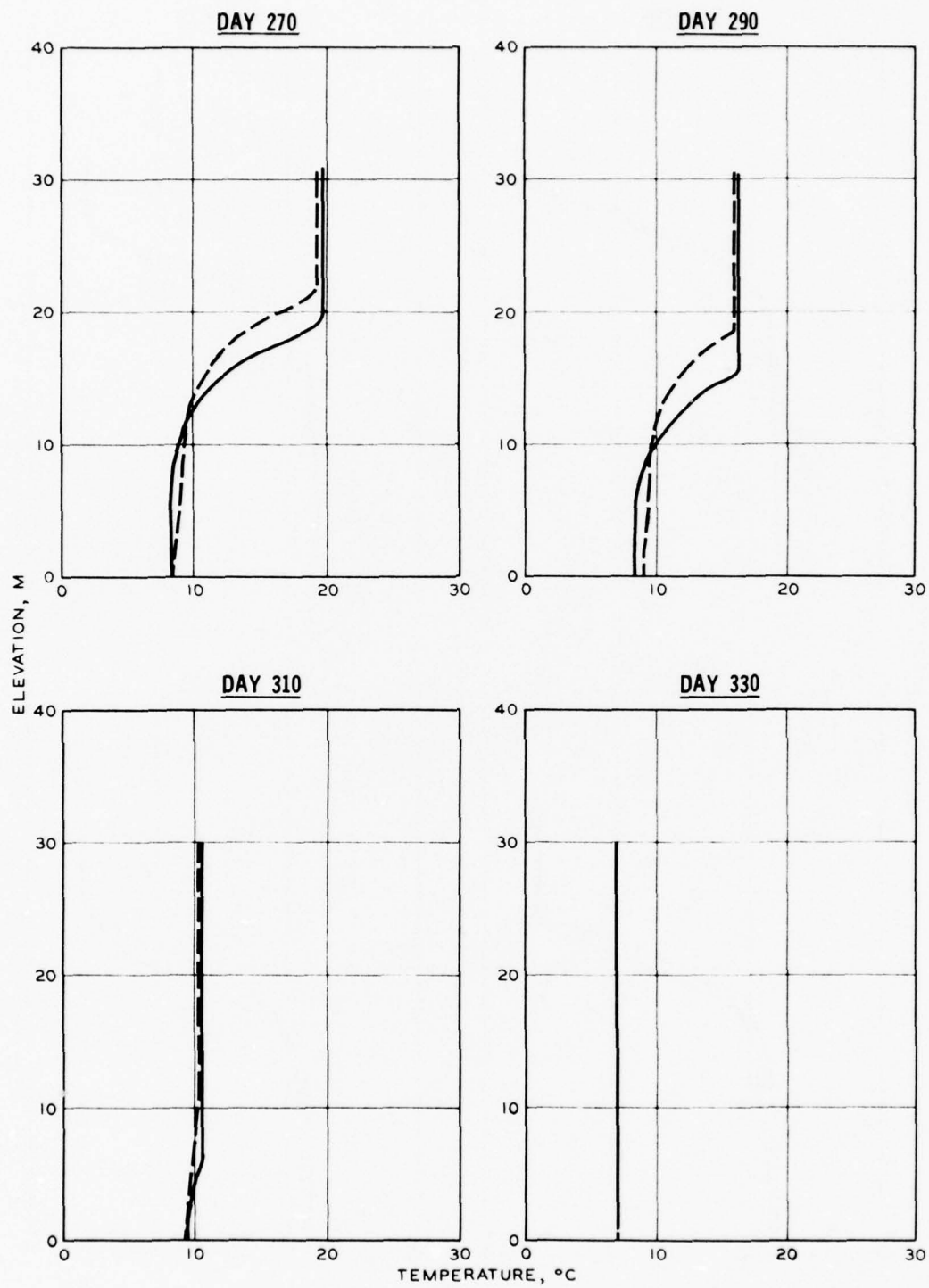


Figure 66 (sheet 3 of 3)

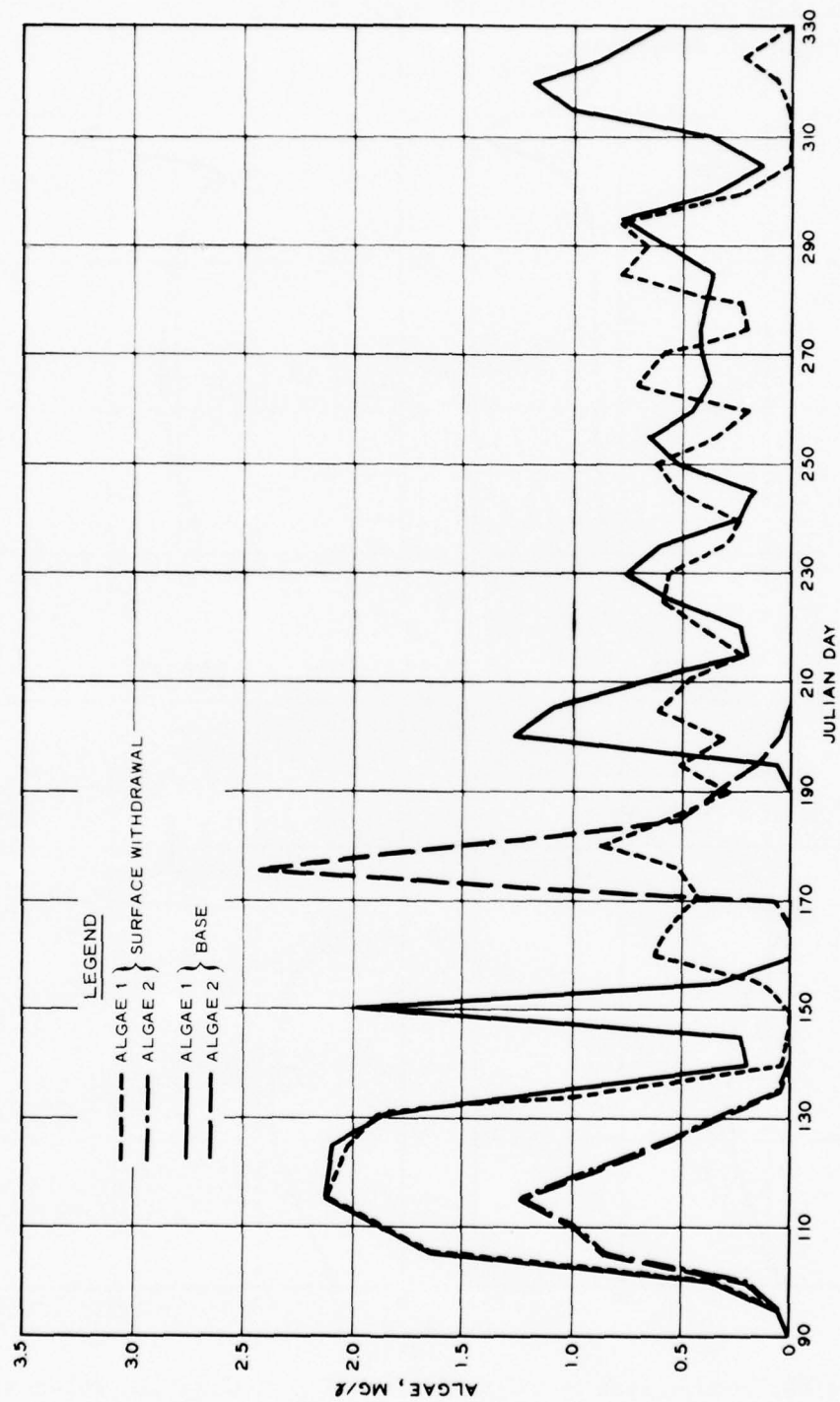


Figure 67. Comparison of algae concentration using surface withdrawal and selective withdrawal (base), 1973

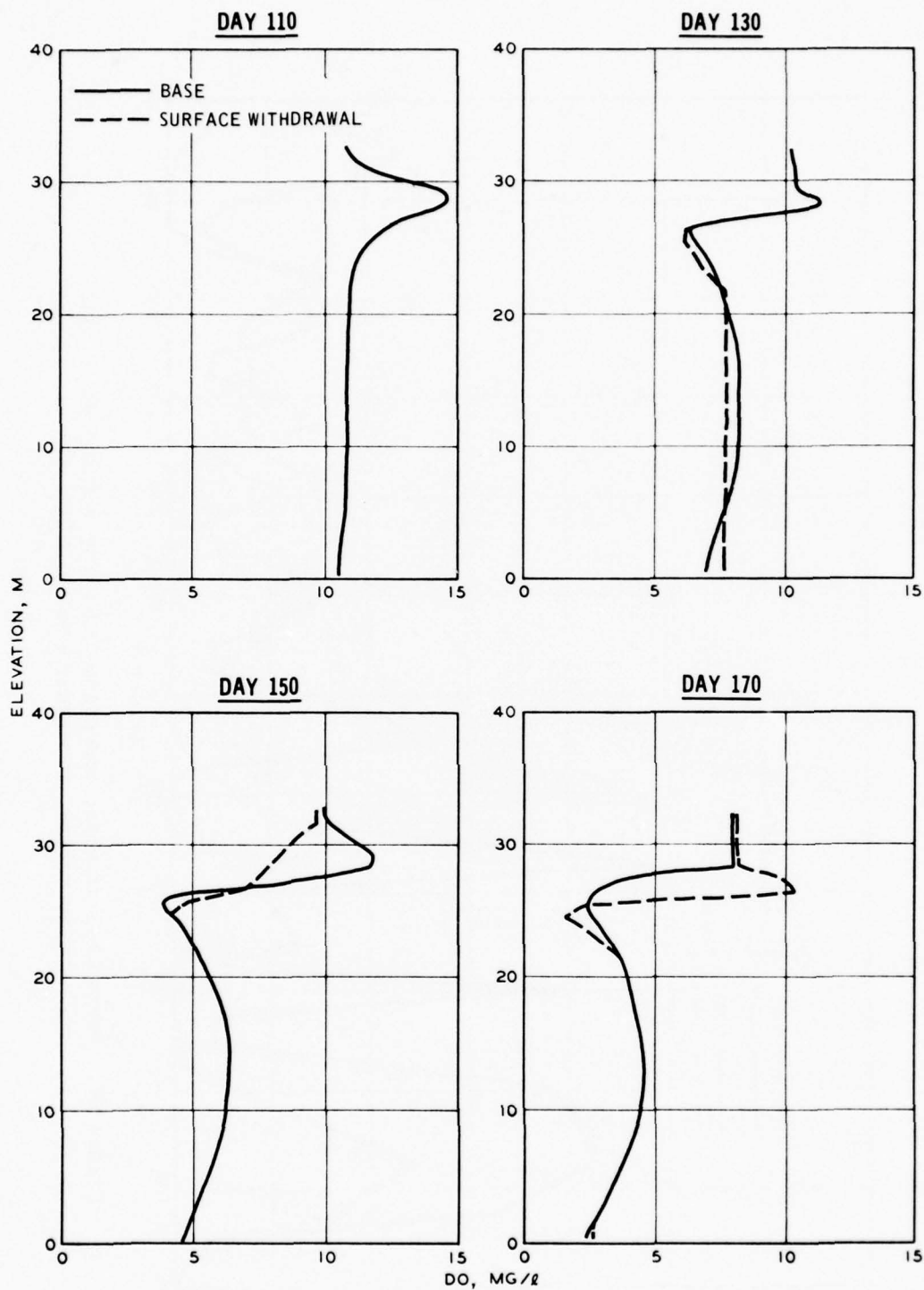


Figure 68. Comparison of DO profiles using surface and selective withdrawal (base), 1973 (sheet 1 of 3)

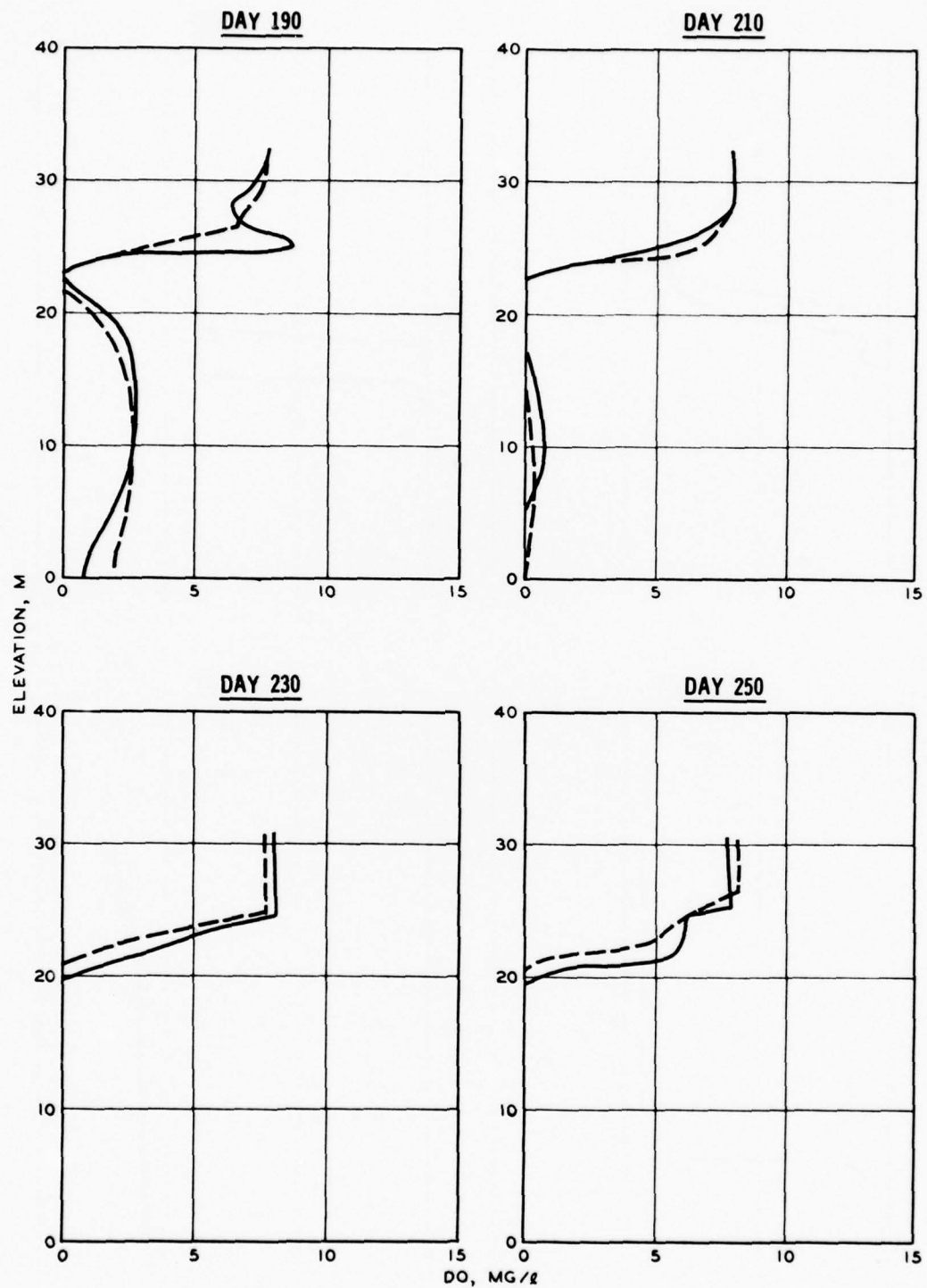


Figure 68 (sheet 2 of 3)

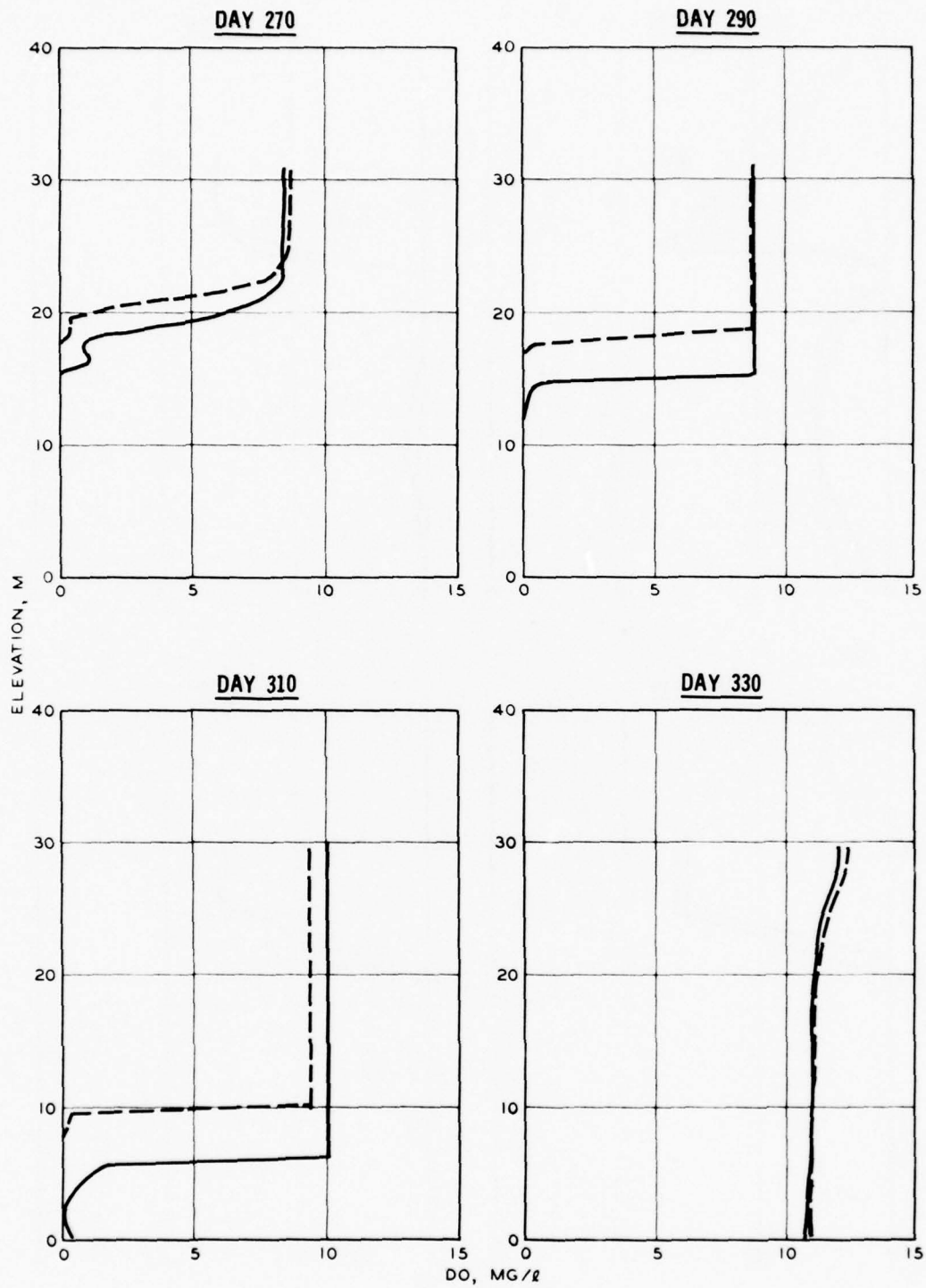


Figure 68 (sheet 3 of 3)

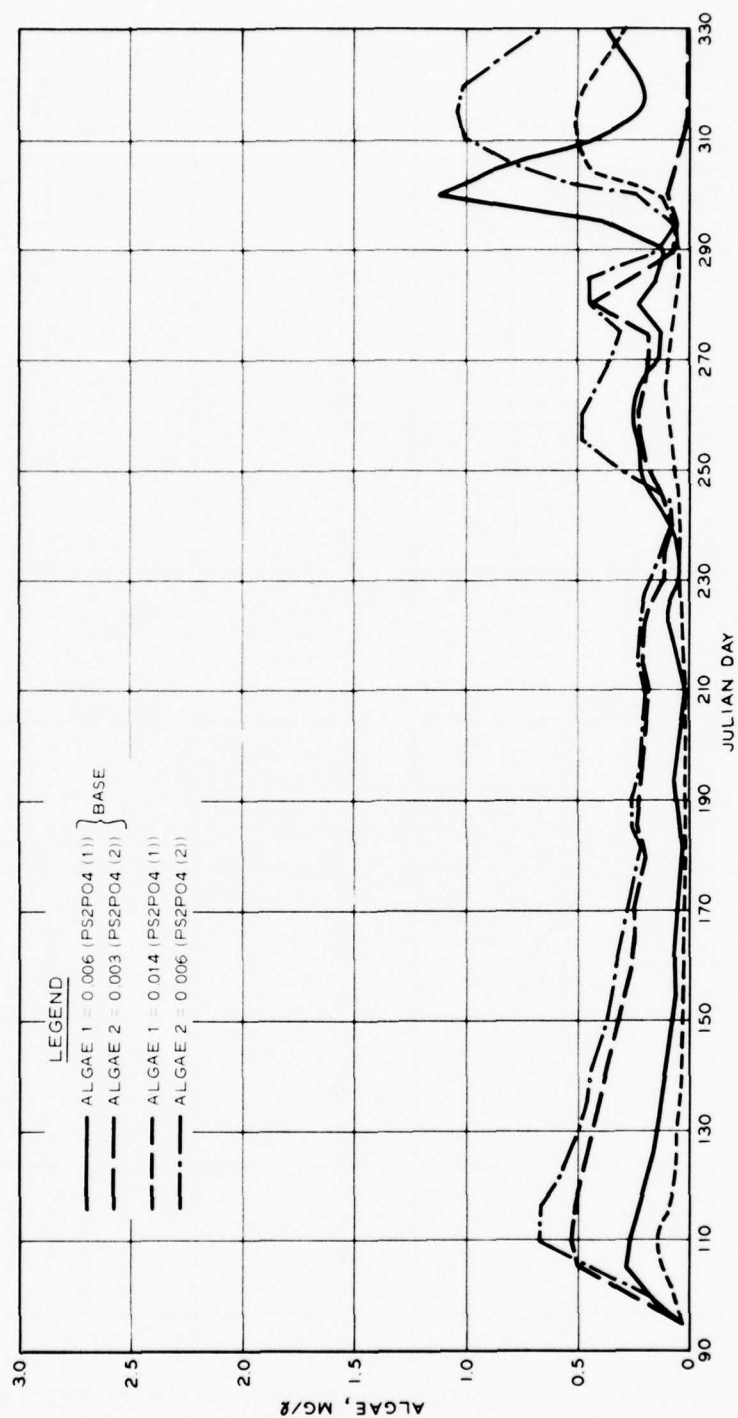


Figure 69. Effect of phosphorus half-saturation coefficients on
ALGAE 1 and 2 concentrations, 1974

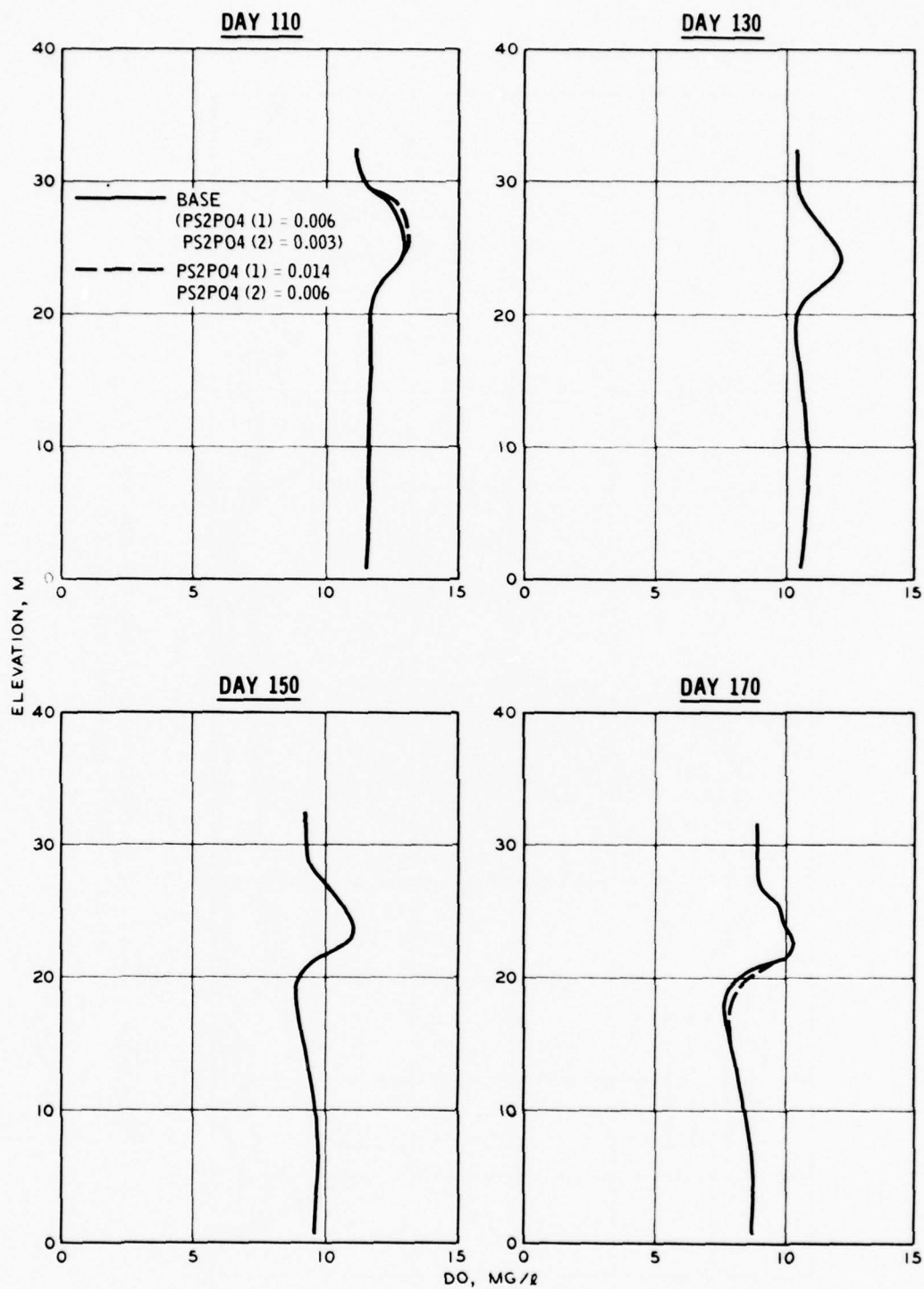


Figure 70. Effect of phosphorus half-saturation coefficients on DO, ALGAE 1 and 2, 1974 (sheet 1 of 3)

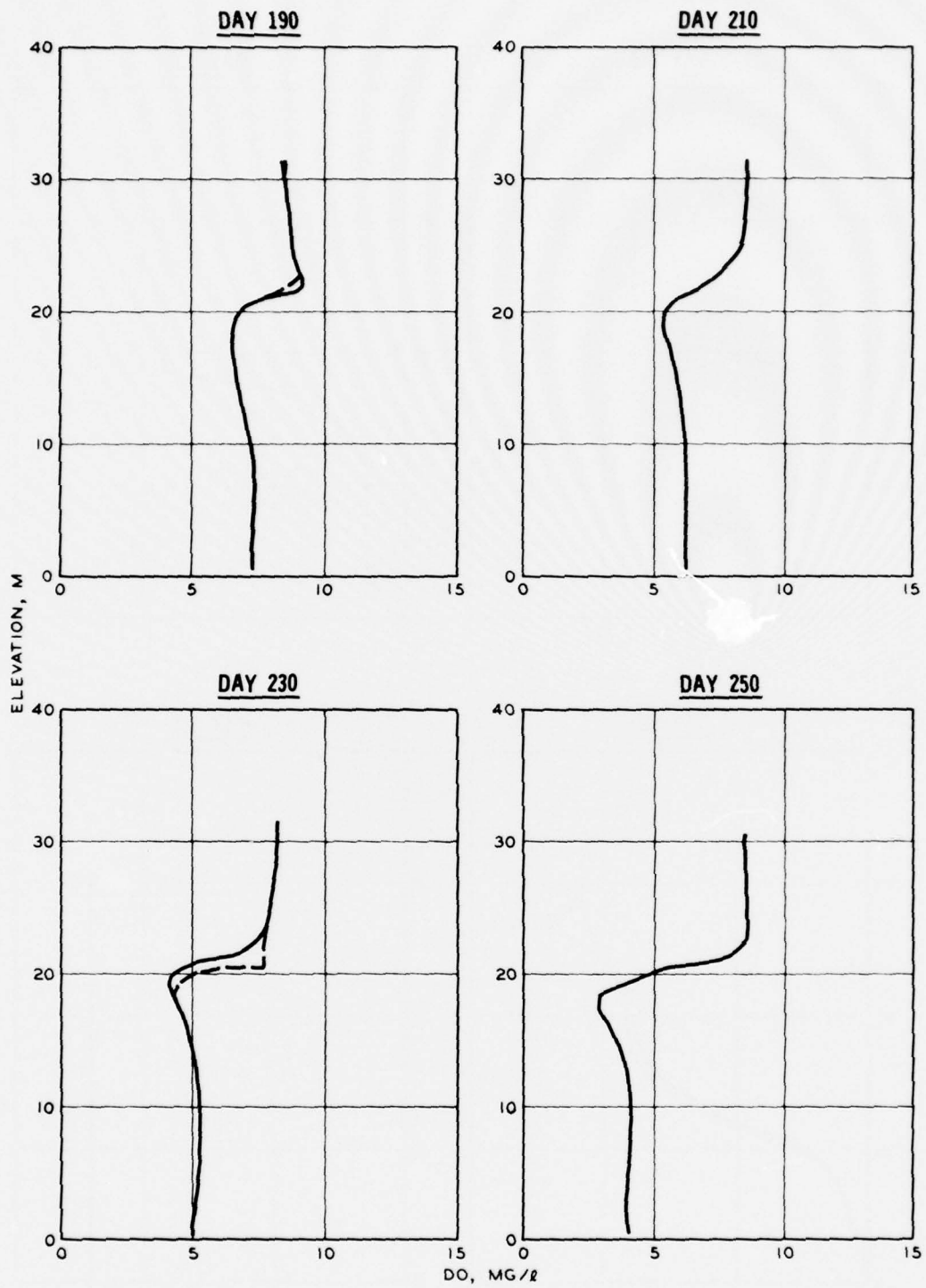


Figure 70 (sheet 2 of 3)

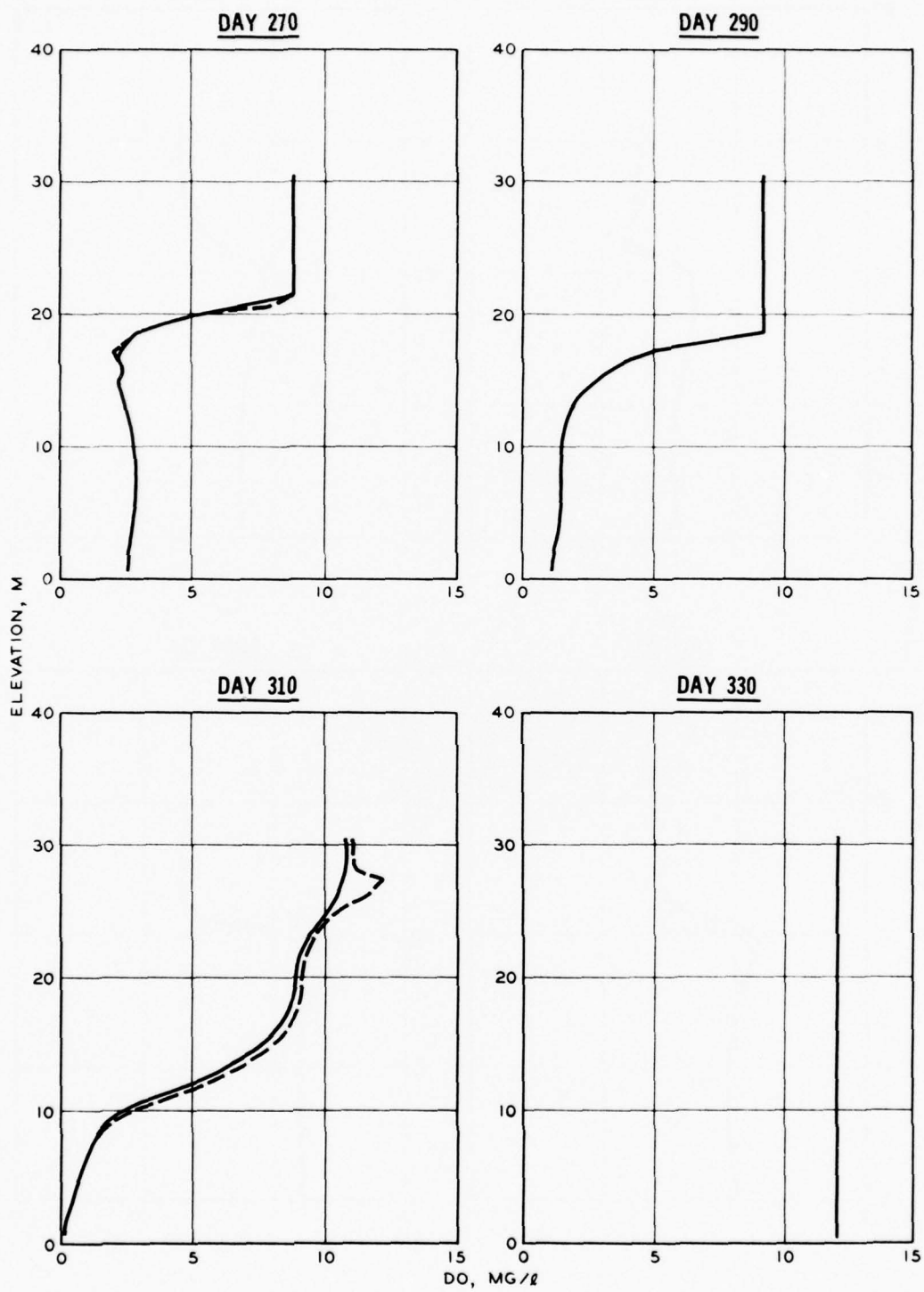


Figure 70 (sheet 3 of 3)

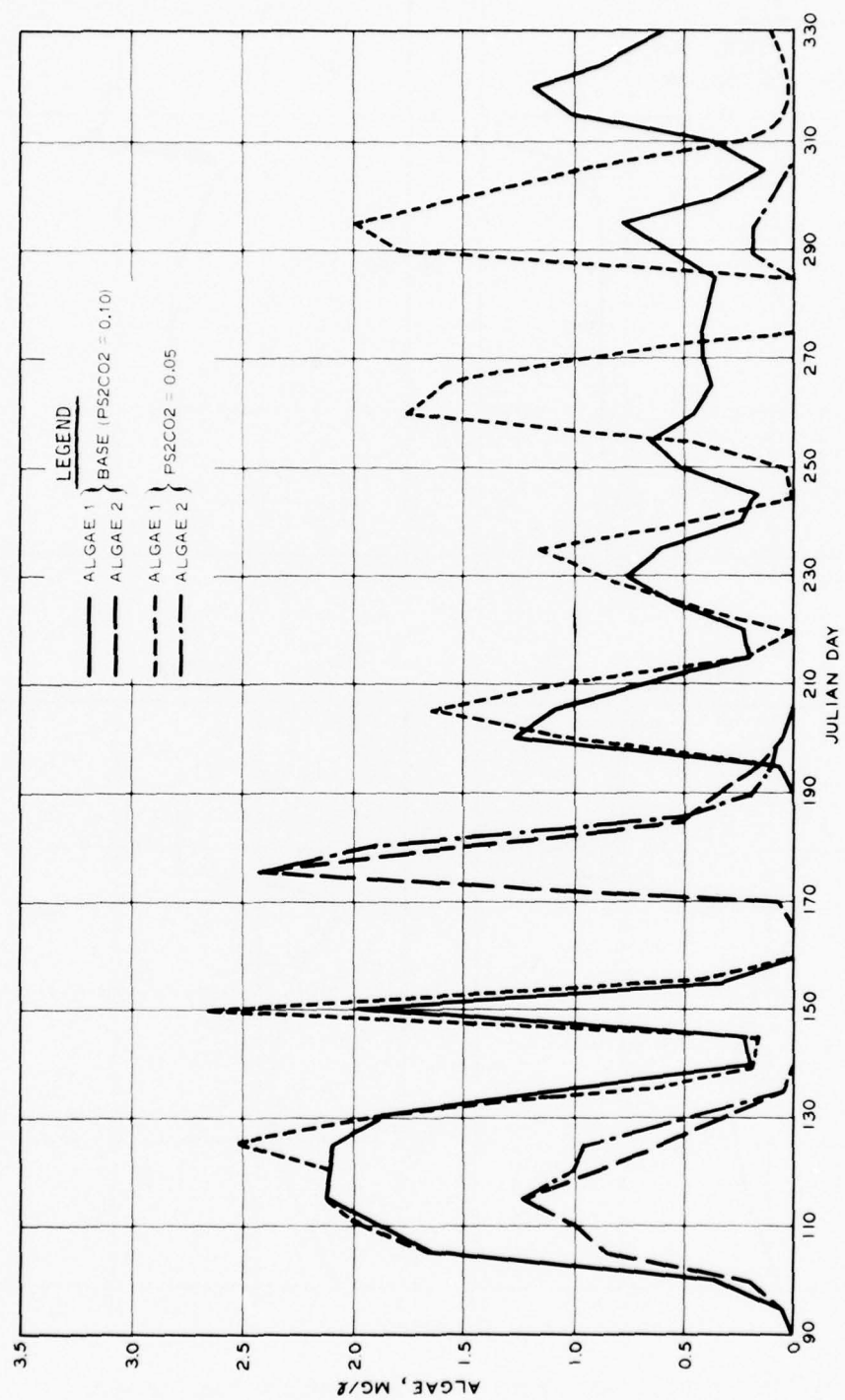


Figure 71. Effect of carbon half-saturation coefficient on ALGAE 1 and 2 concentrations, 1973

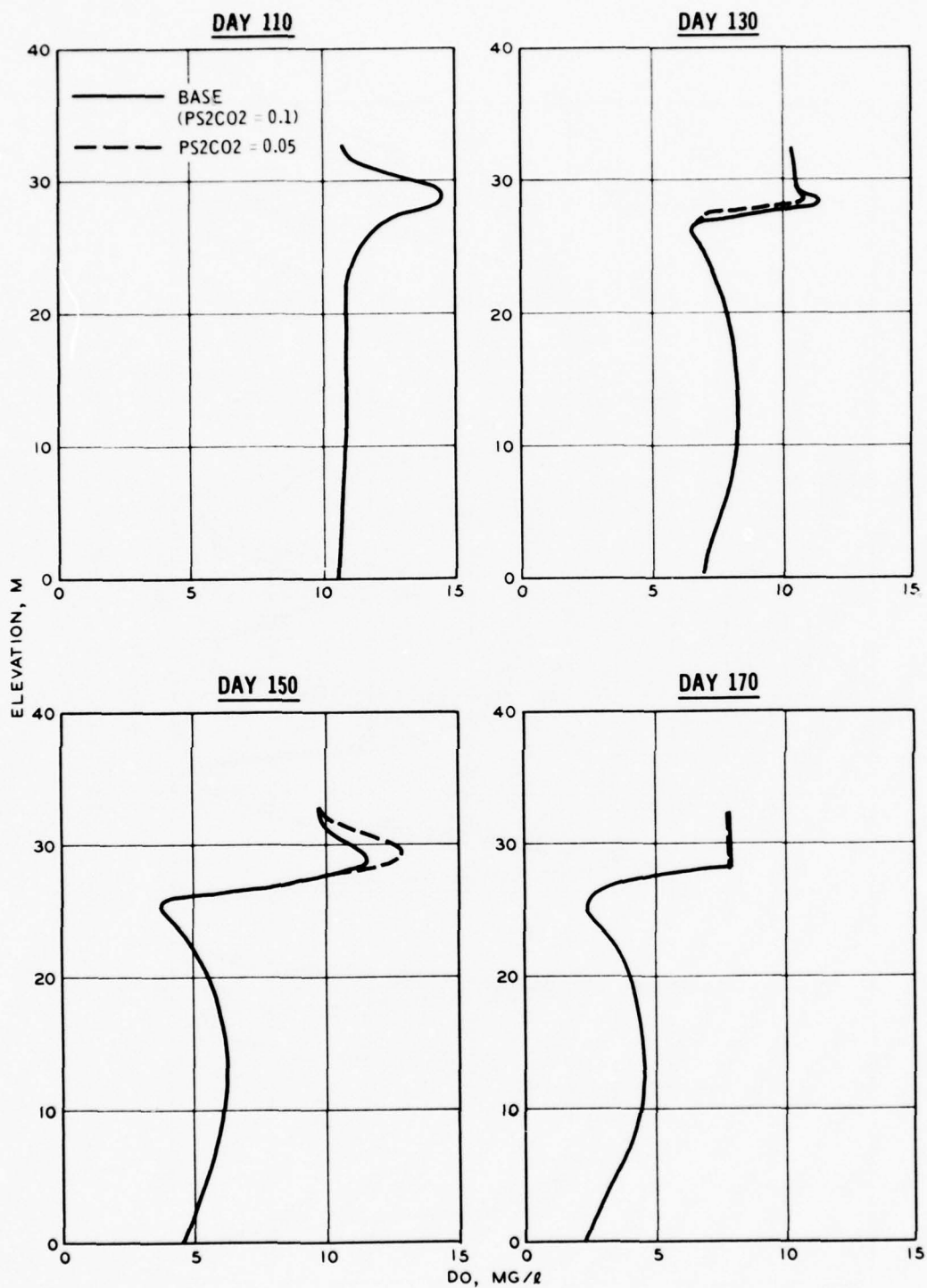


Figure 72. Effect of carbon half-saturation coefficients on DO (sheet 1 of 3)

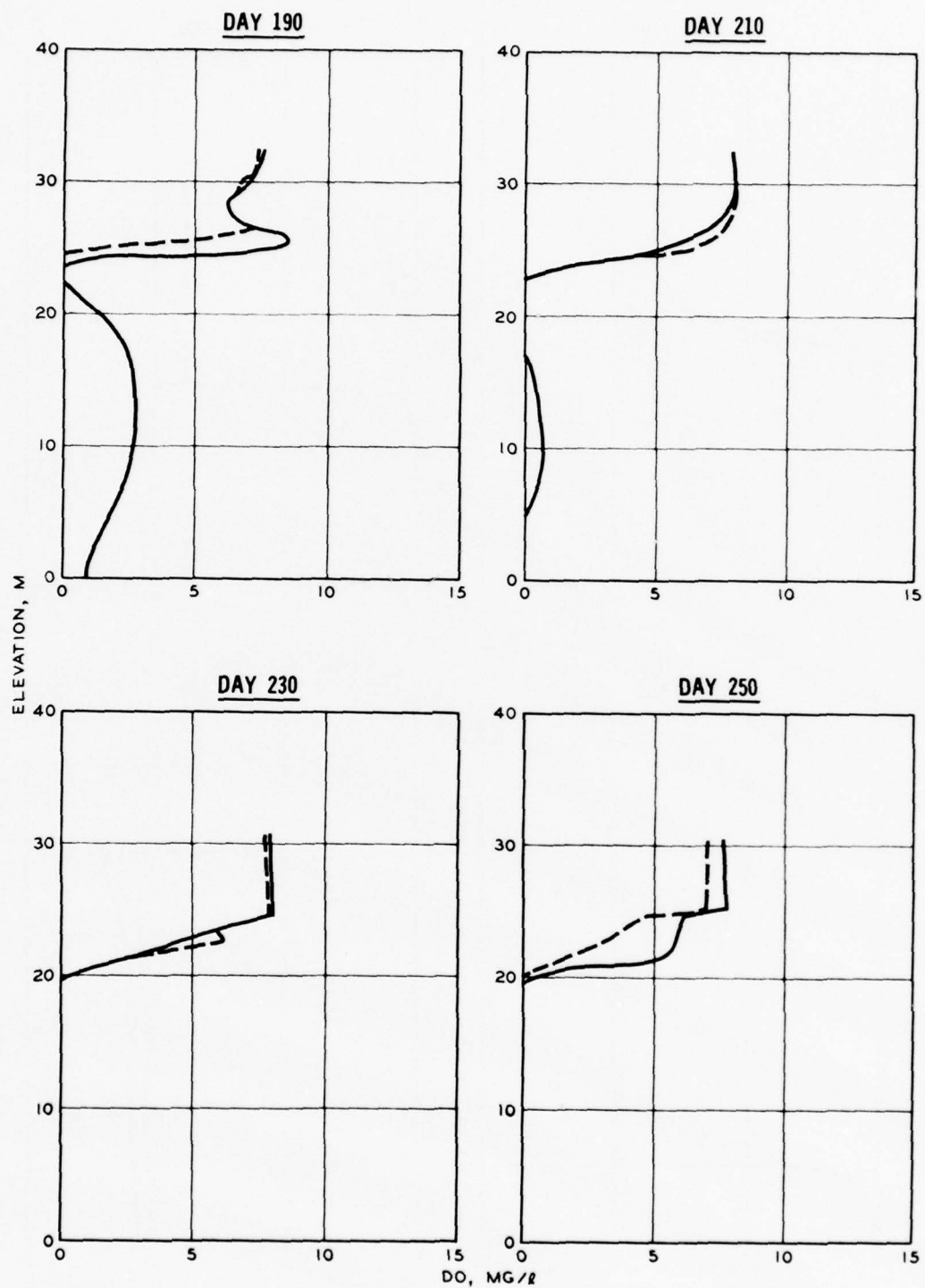


Figure 72 (sheet 2 of 3)

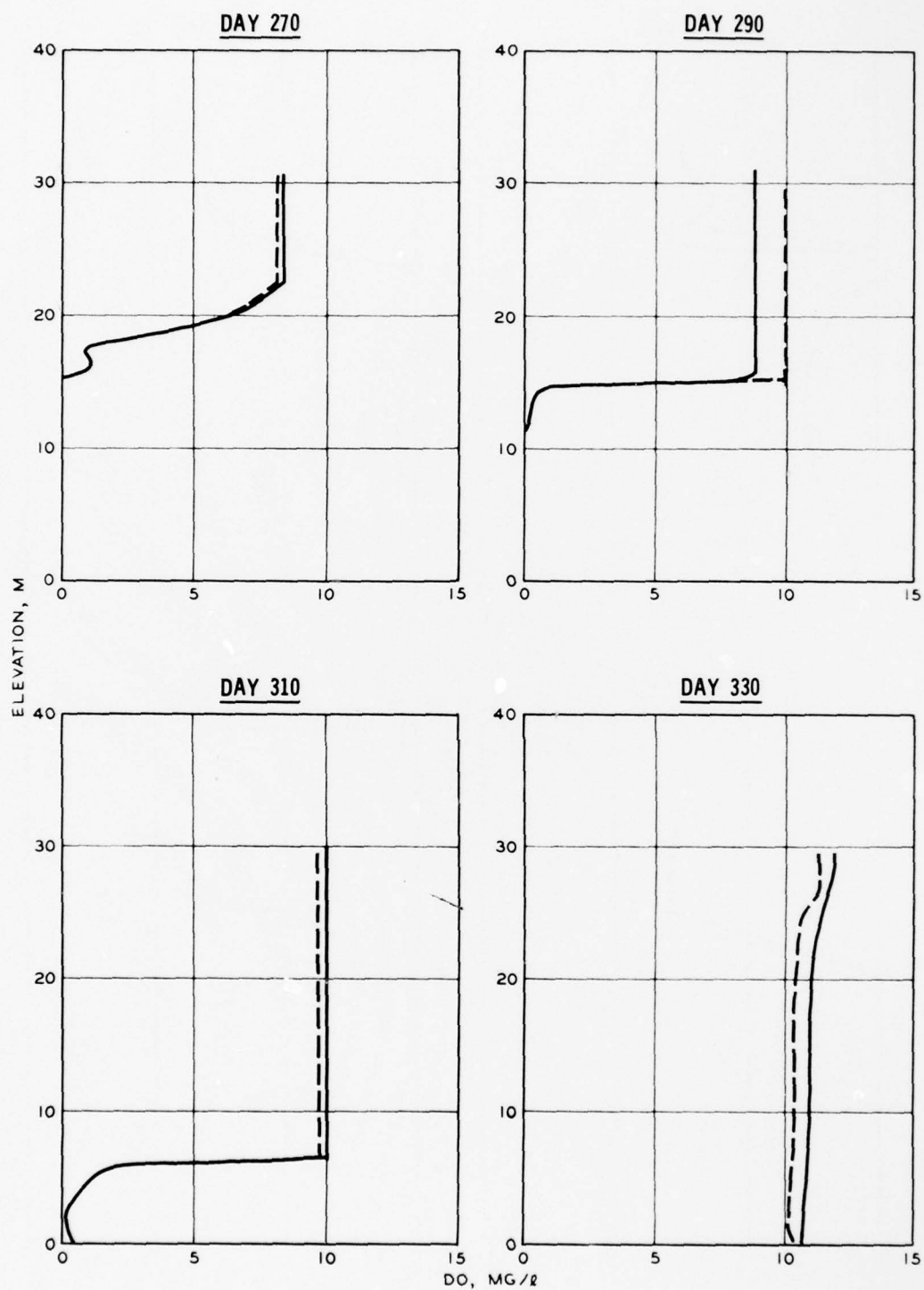


Figure 72 (sheet 3 of 3)

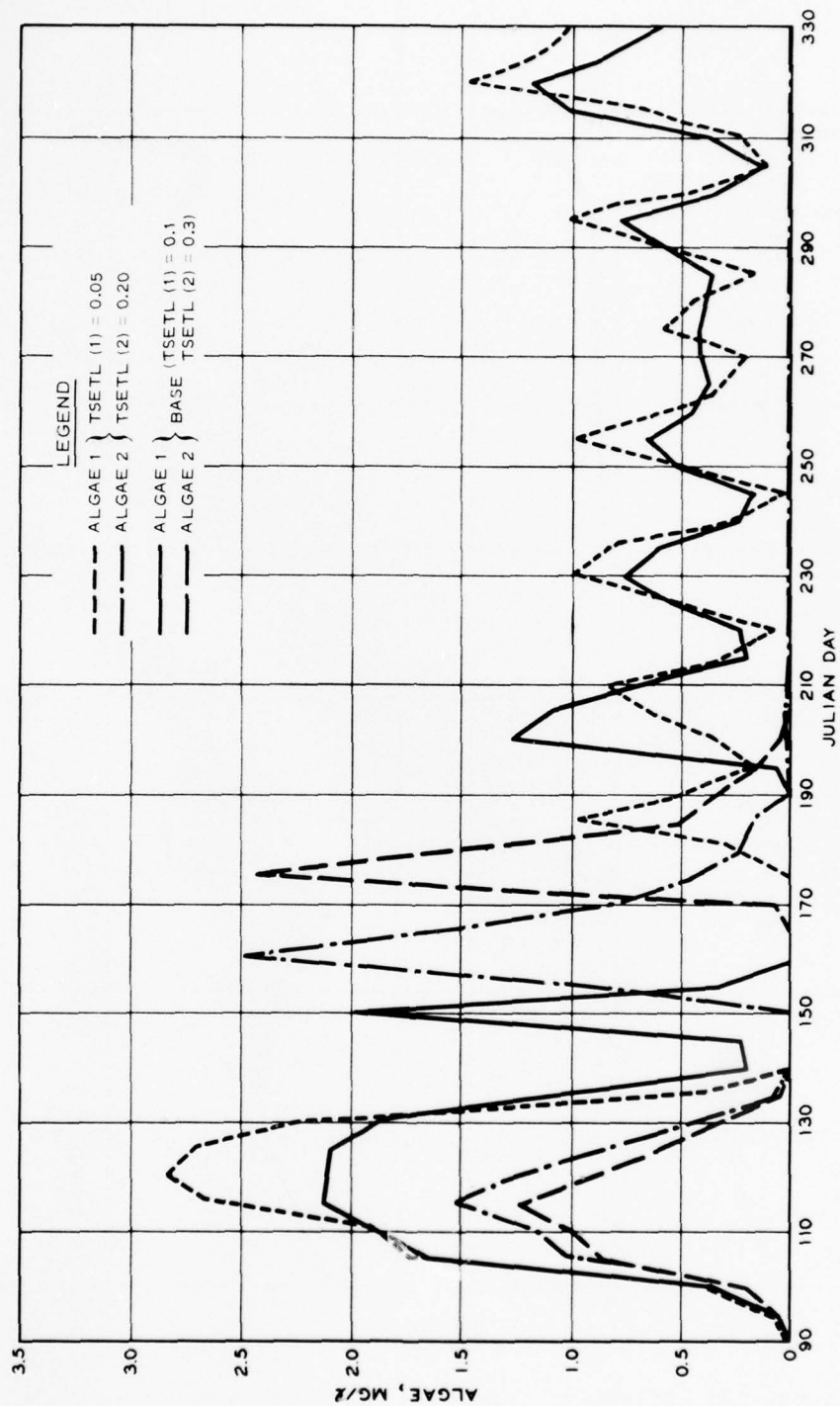


Figure 73. Effect of decreased settling velocity on ALGAE 1 and 2 concentrations, 1973

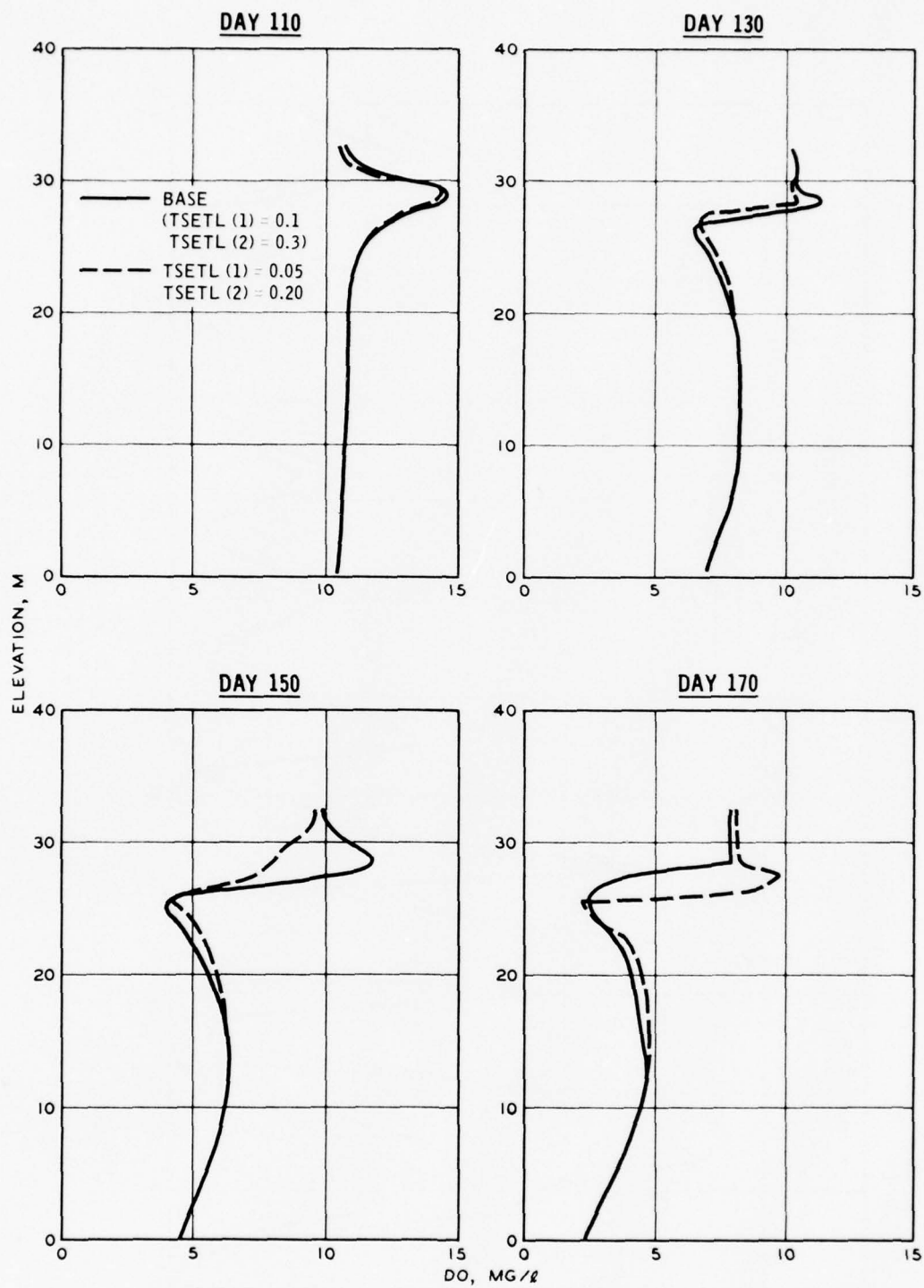


Figure 74. Effect of decreased settling velocity on DO, 1973 (sheet 1 of 3)

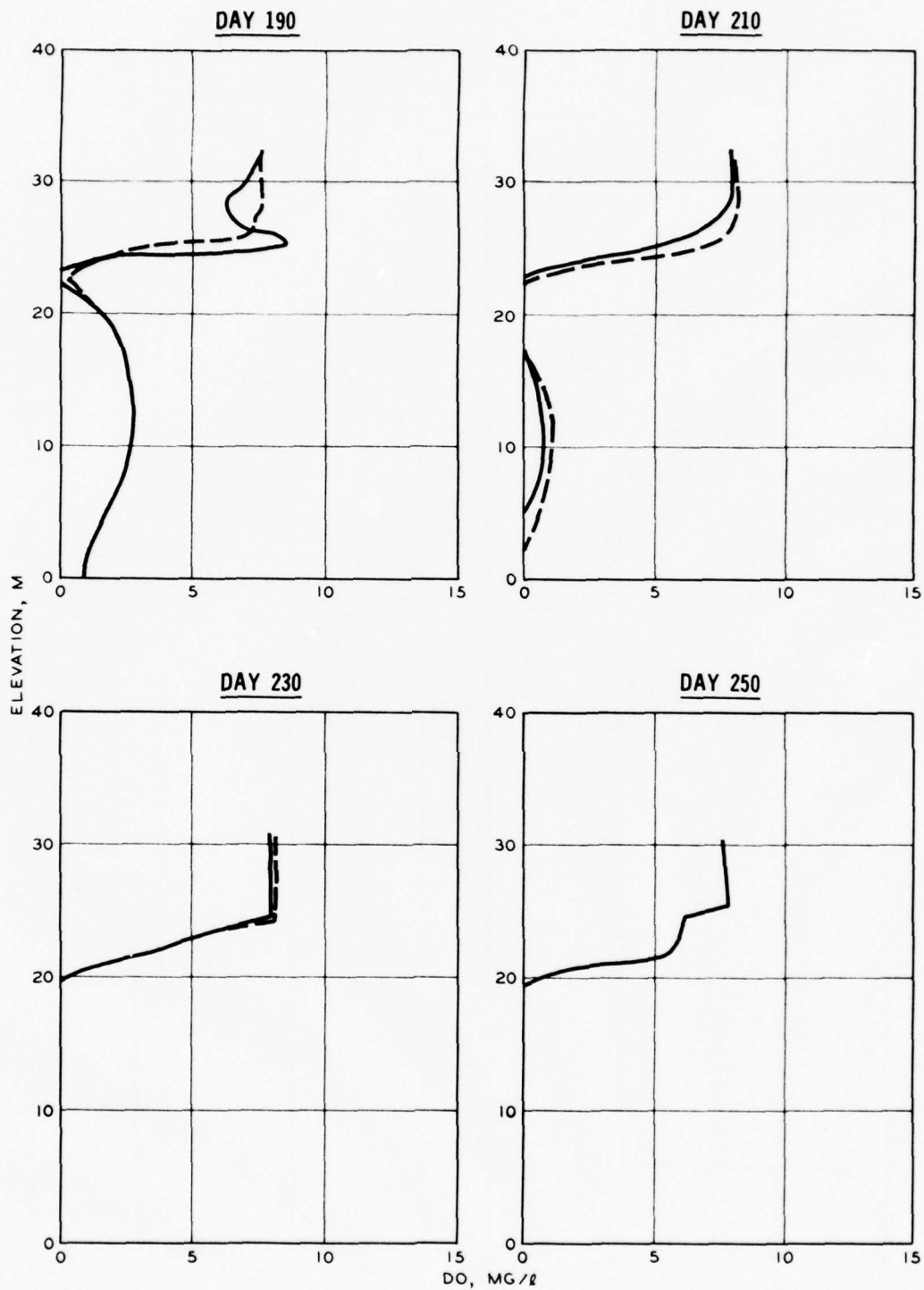


Figure 74 (sheet 2 of 3)

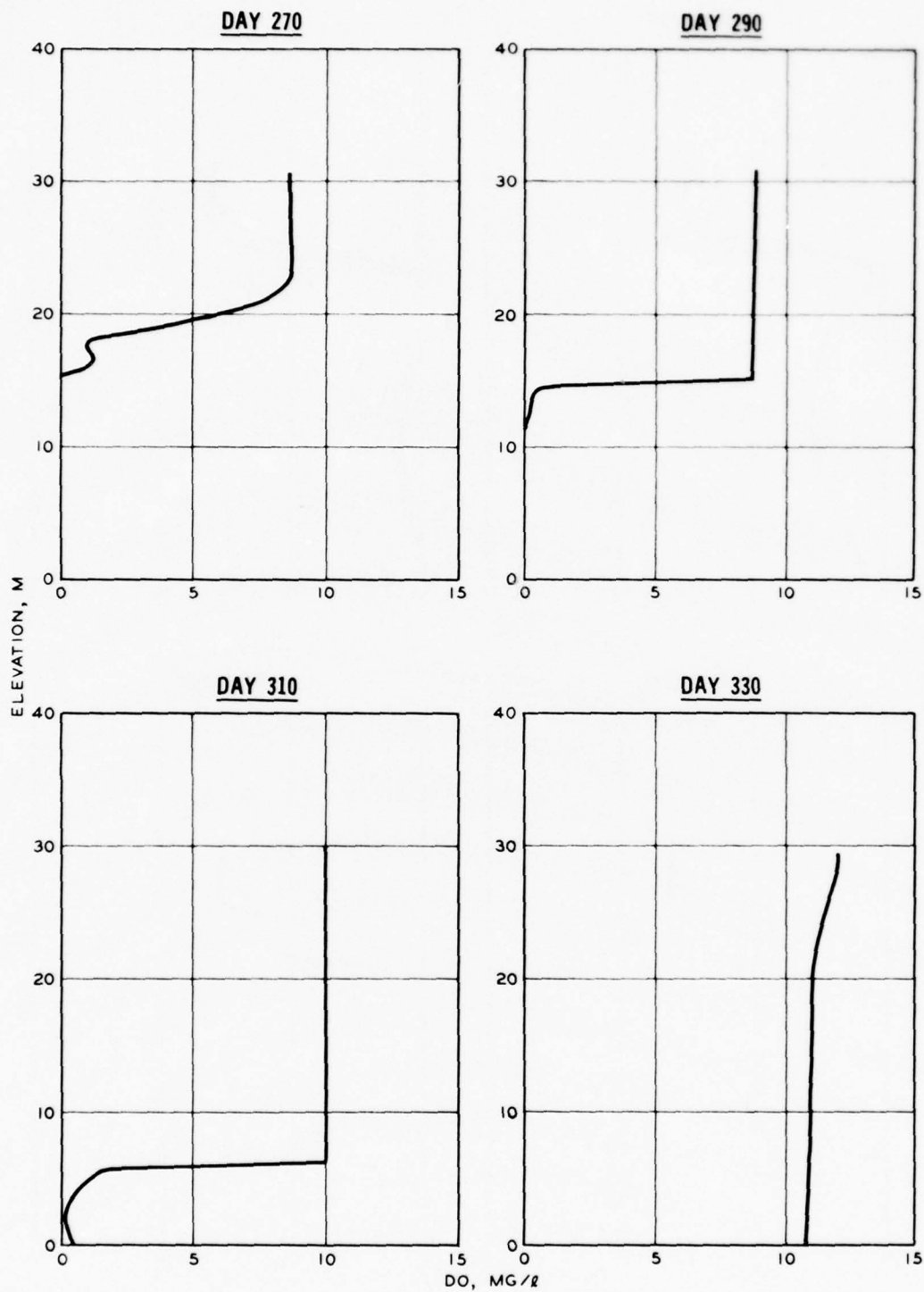


Figure 74 (sheet 3 of 3)

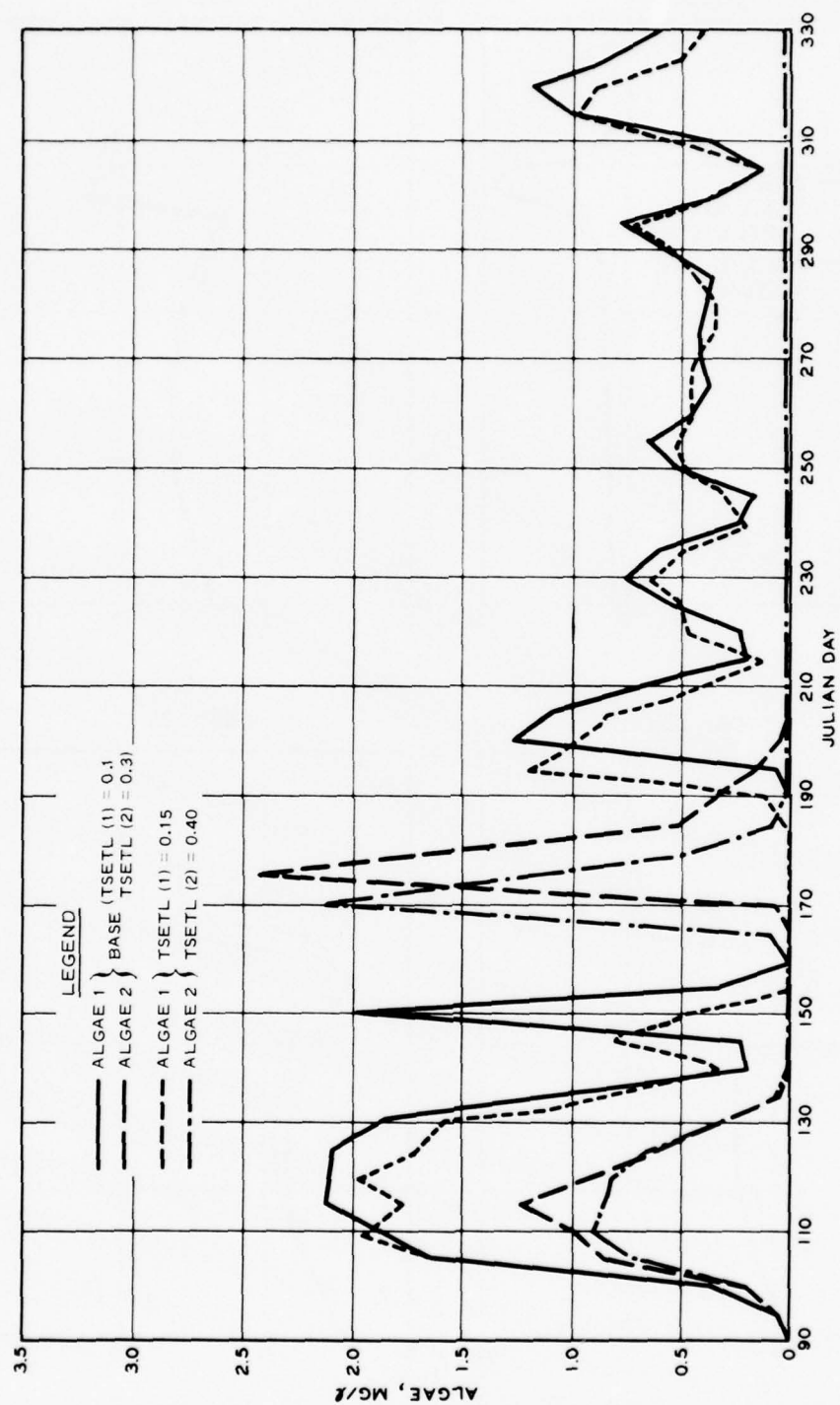


Figure 75. Effect of increased settling velocity on ALGAE 1 and 2 concentrations, 1973

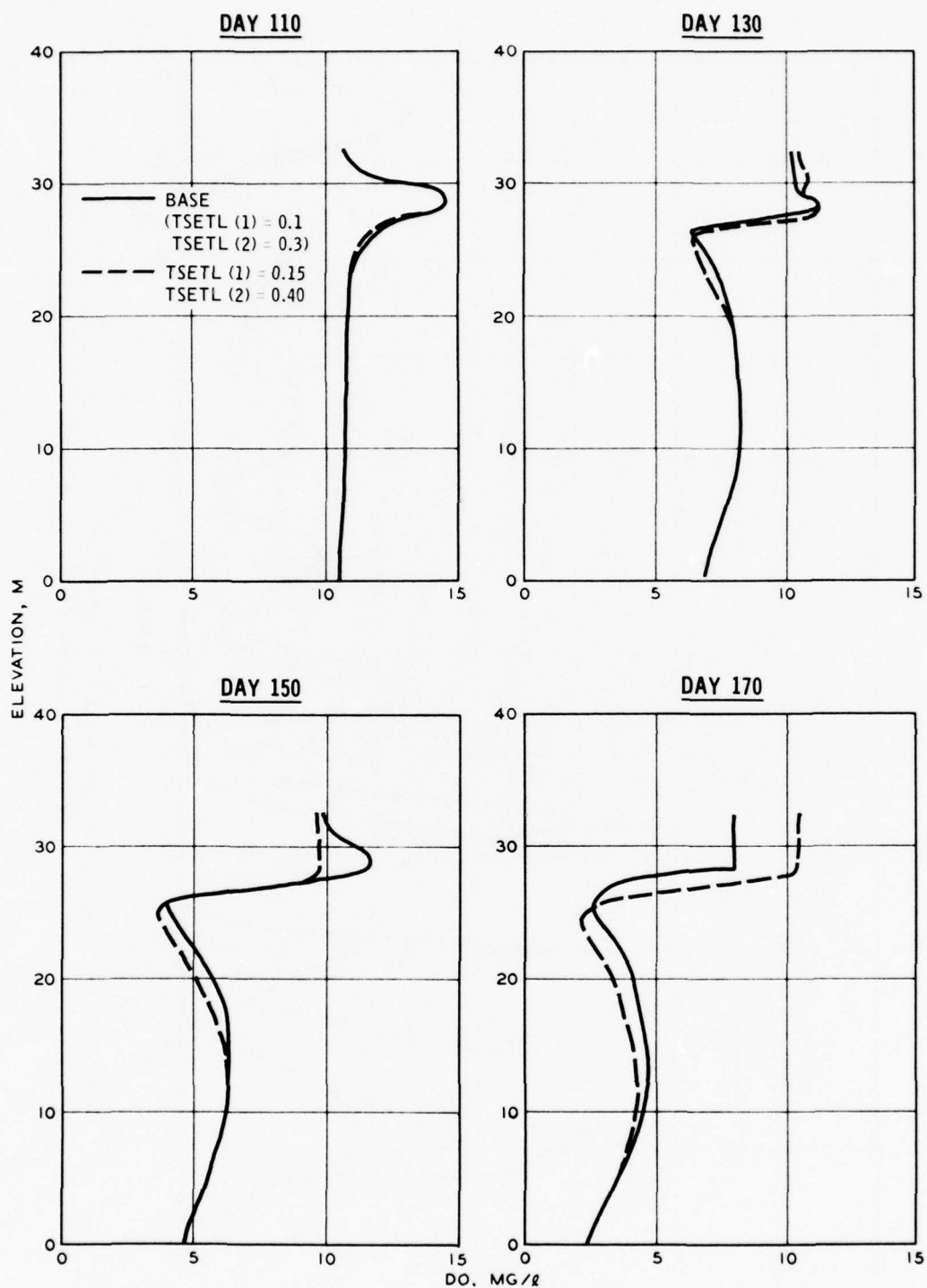


Figure 76. Effect of increased settling velocity on DO, 1973 (sheet 1 of 3)

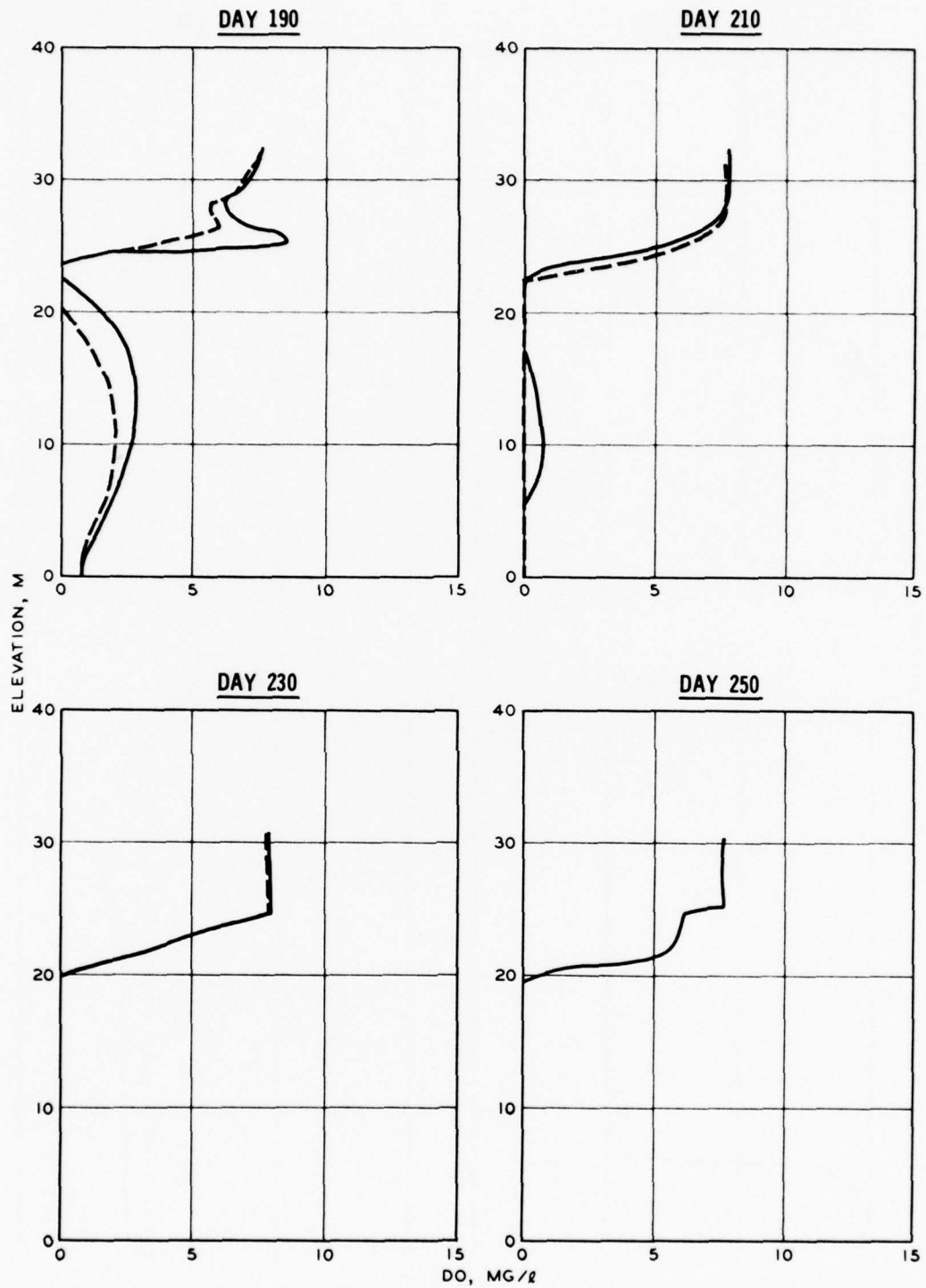


Figure 76 (sheet 2 of 3)

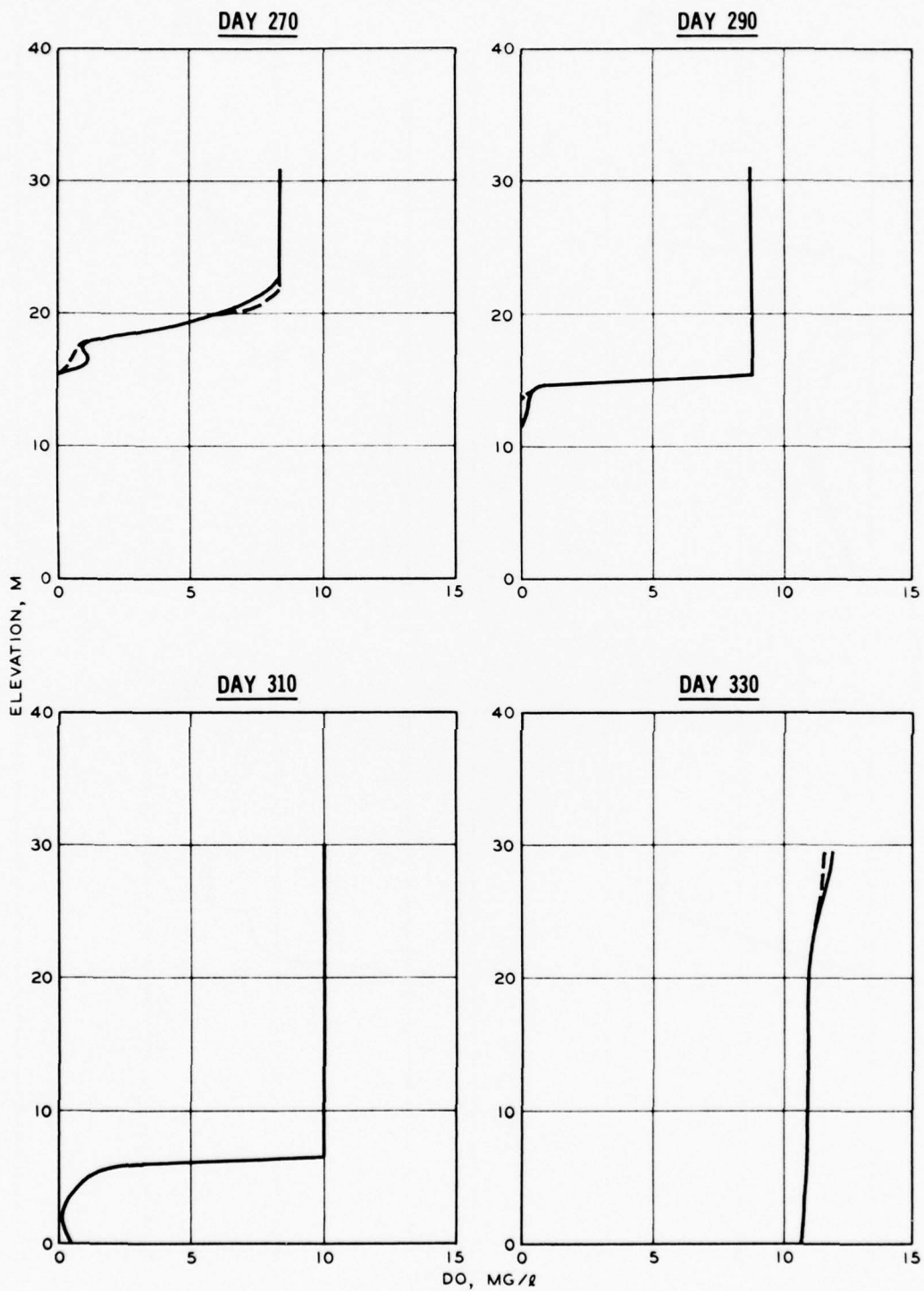


Figure 76 (sheet 3 of 3)

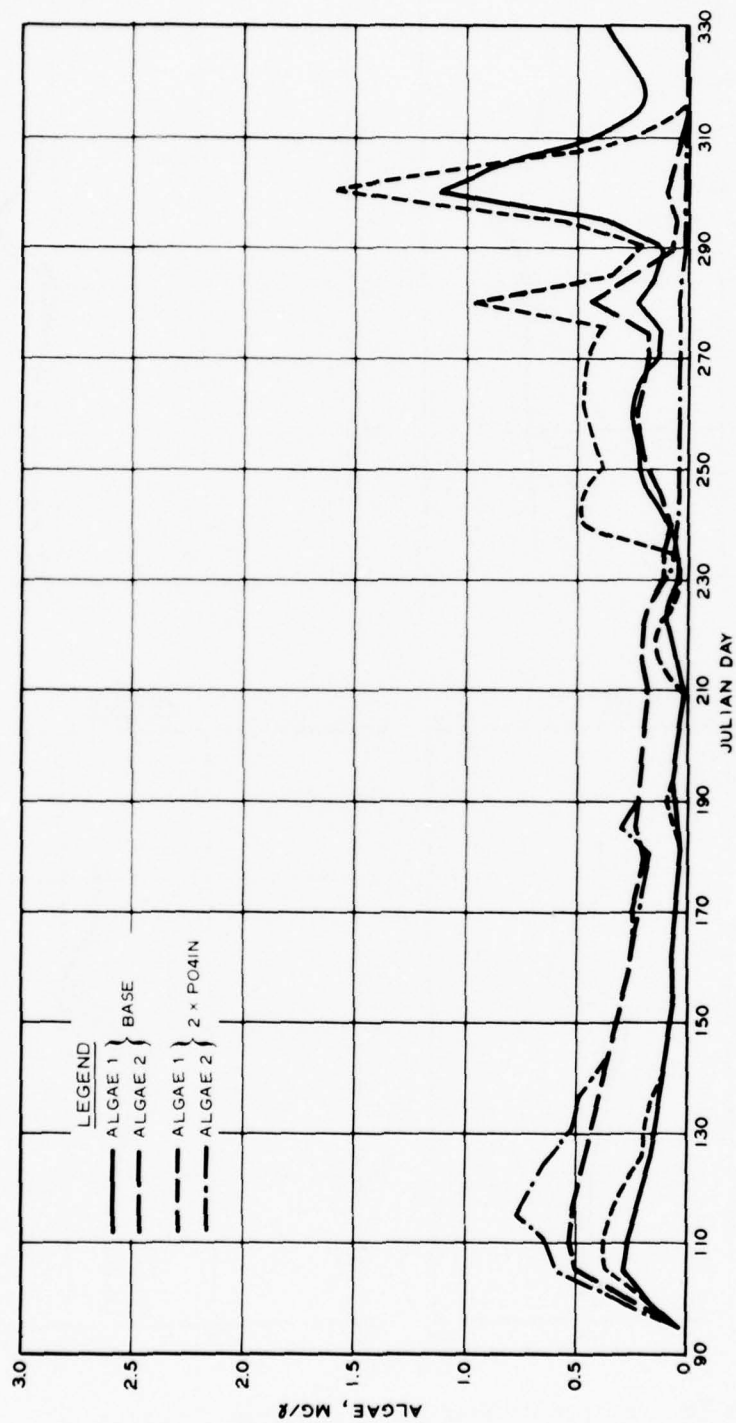


Figure 77. Effect of doubled phosphorus loadings on ALGAE 1 and 2 concentrations, 1974

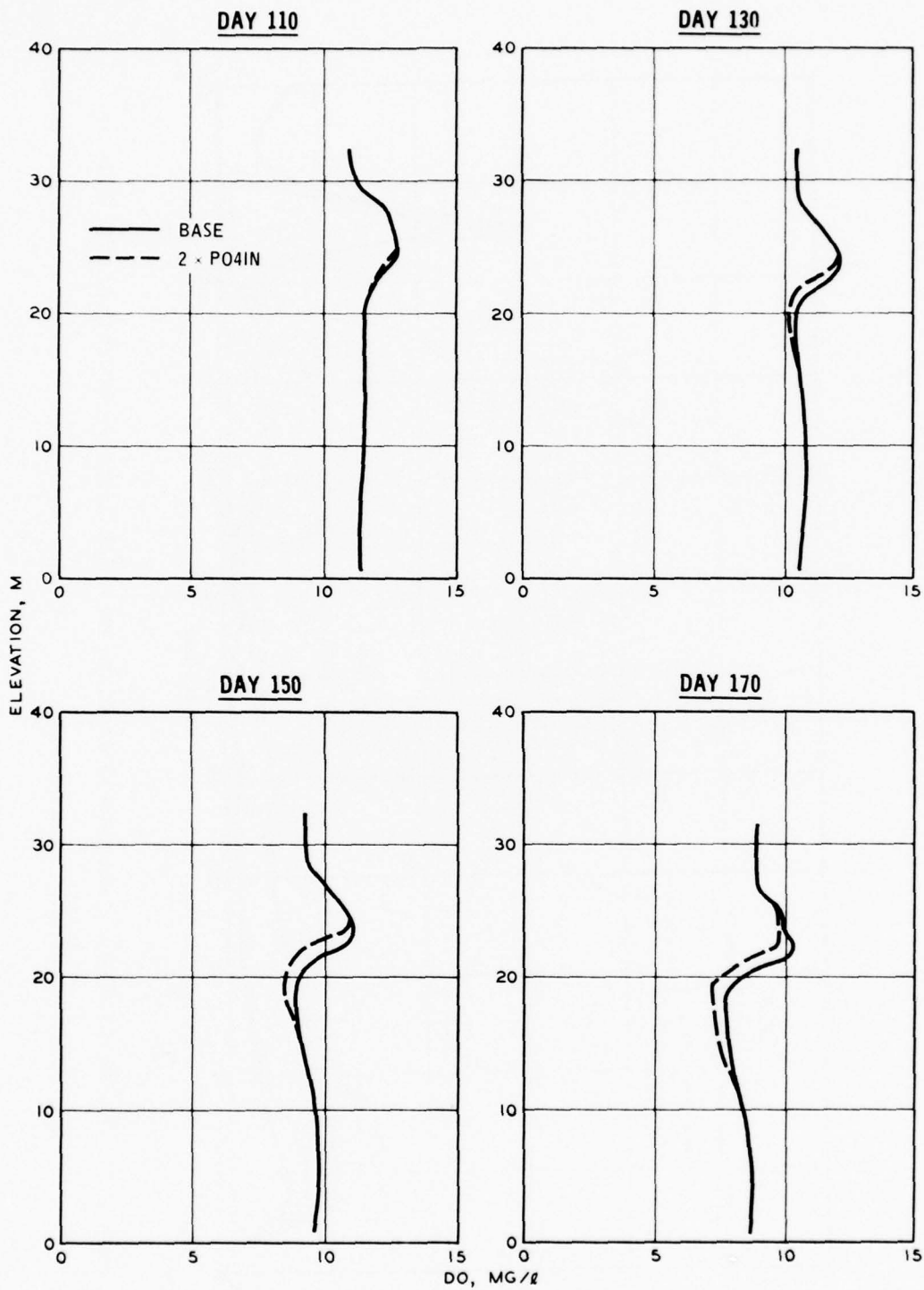


Figure 78. Effect of doubled phosphorus loadings on DO concentrations, 1974 (sheet 1 of 3)

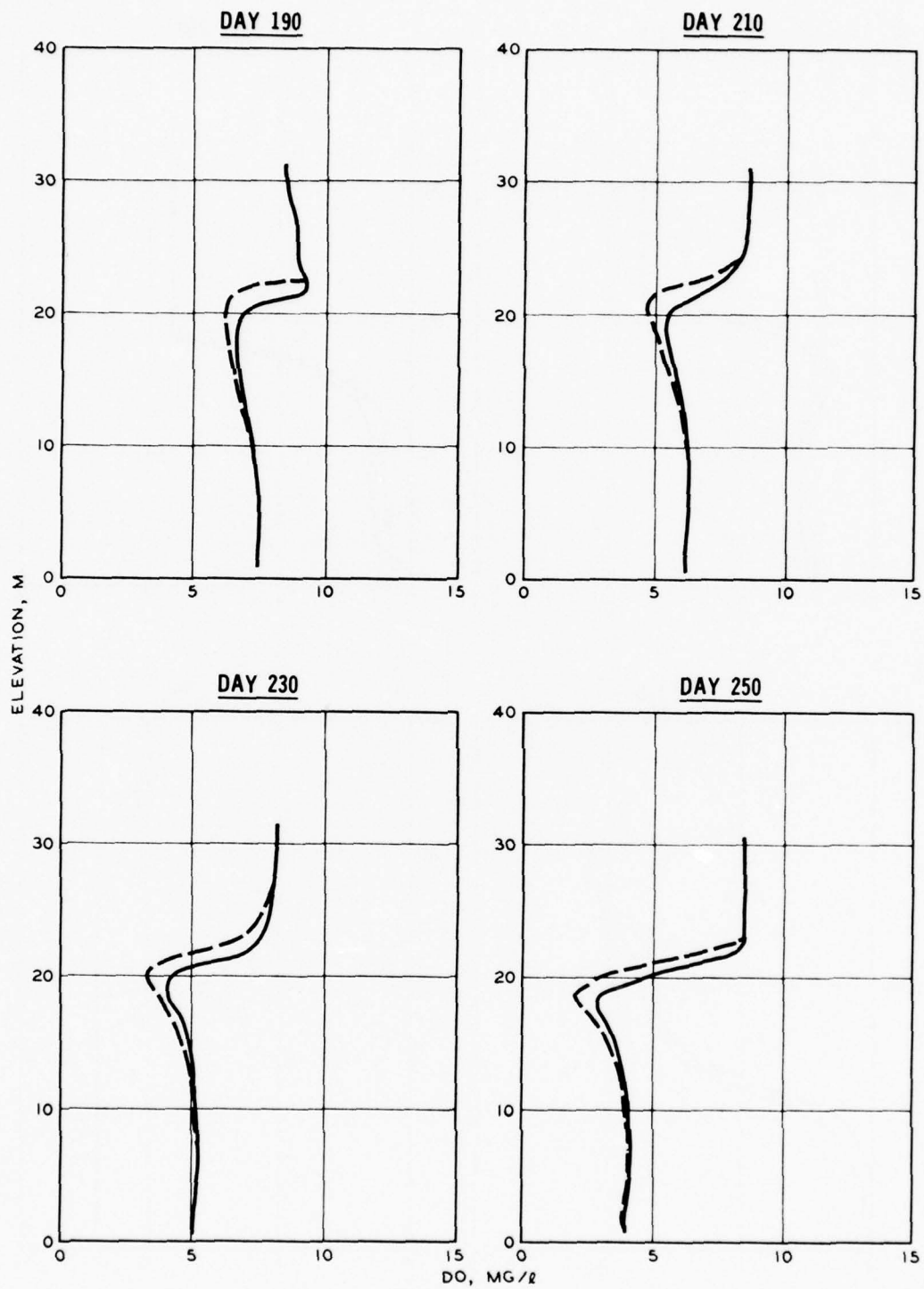


Figure 78 (sheet 2 of 3)

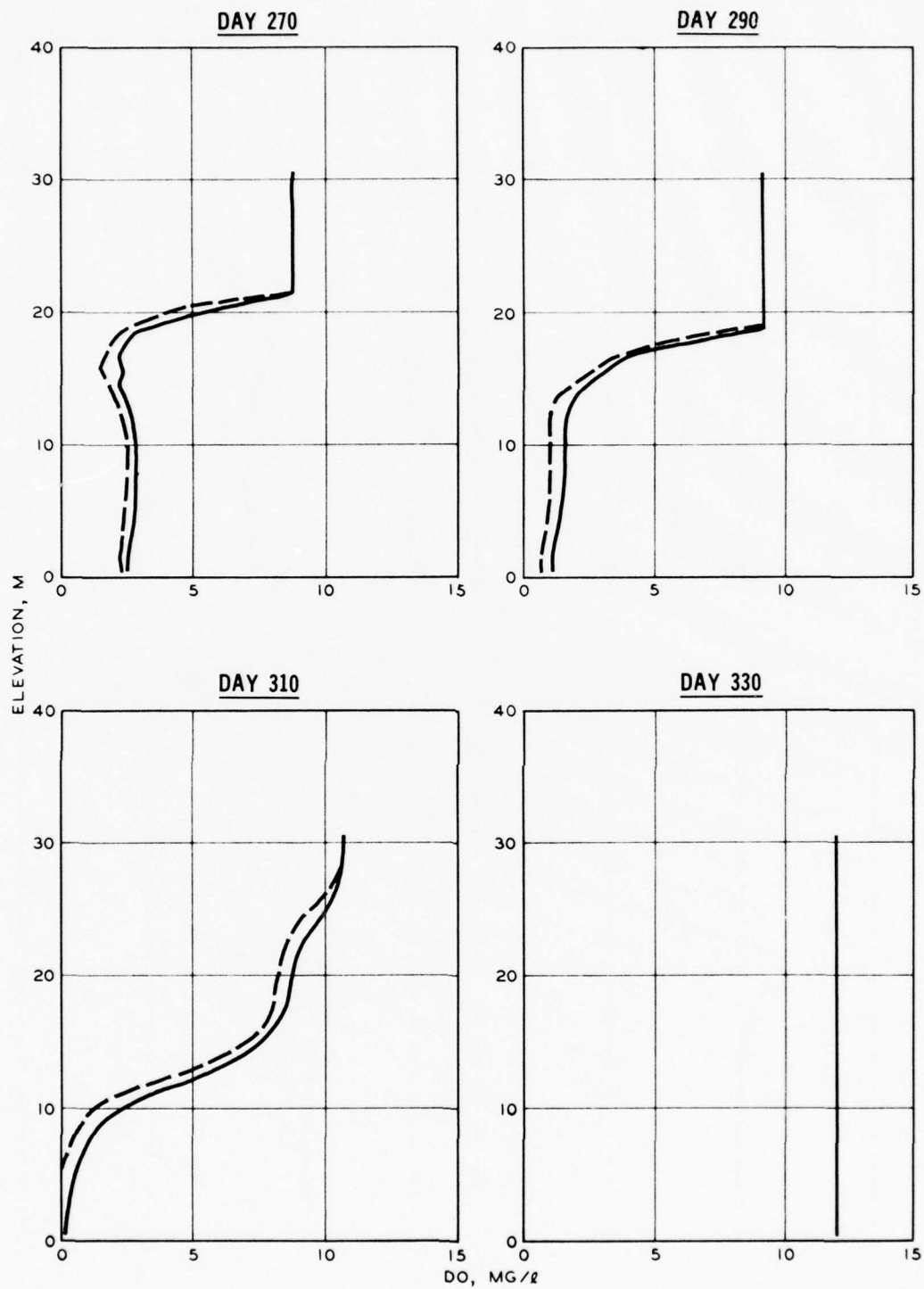


Figure 78 (sheet 3 of 3)

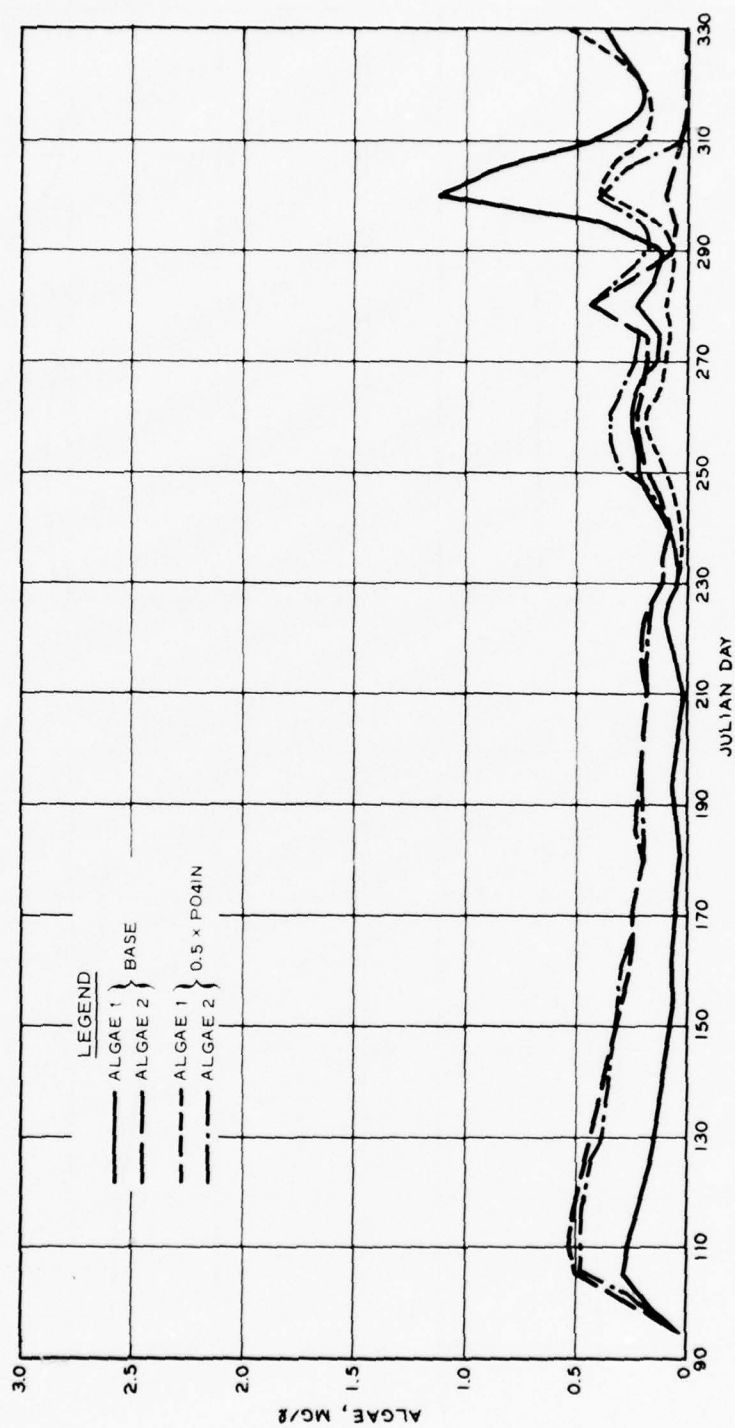


Figure 79. Effect of halved phosphorus loadings on ALGAE 1 and 2 concentrations, 1974

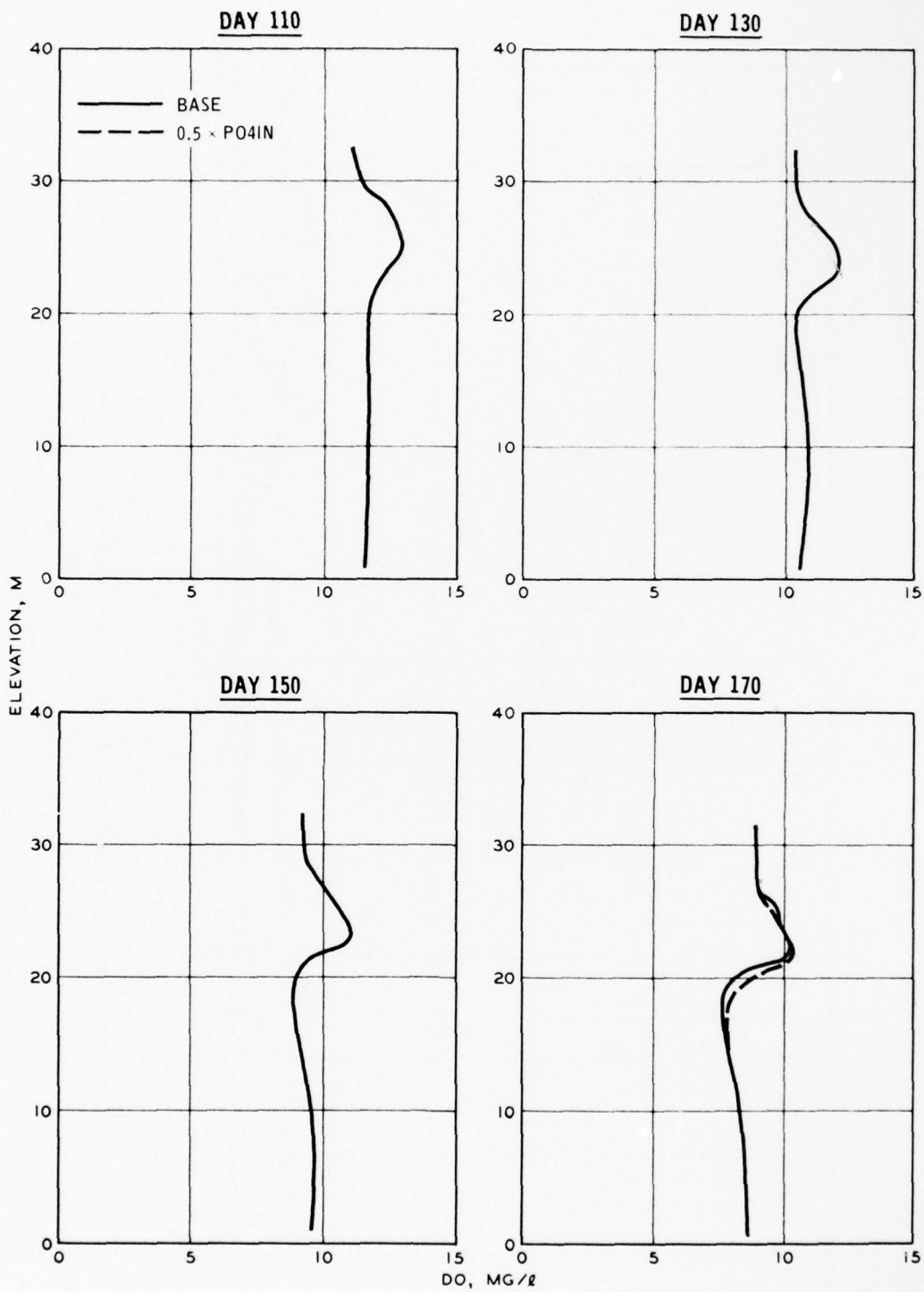


Figure 80. Effect of halved phosphorus loadings on DO, 1974 (sheet 1 of 3)

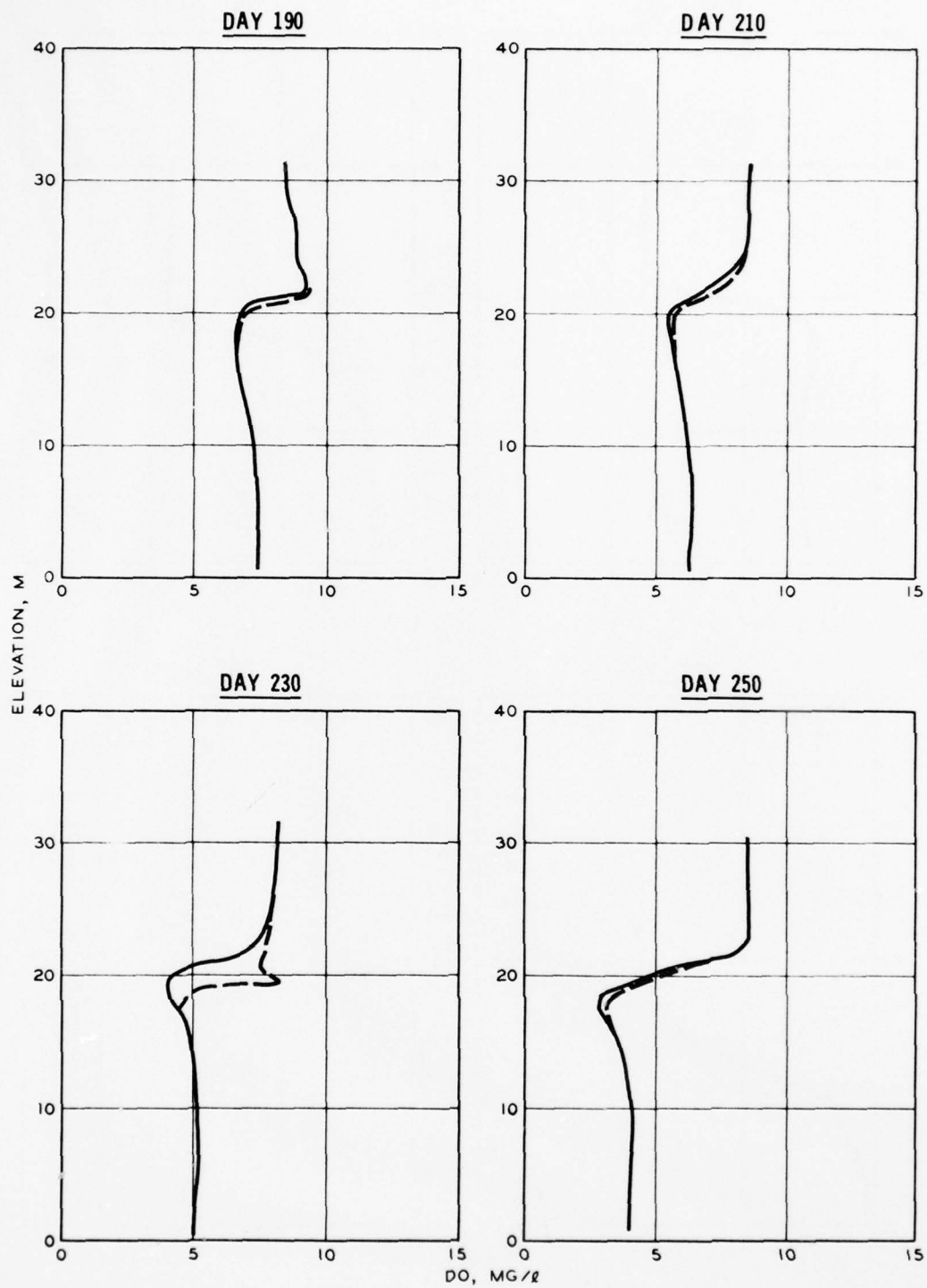


Figure 80 (sheet 2 of 3)

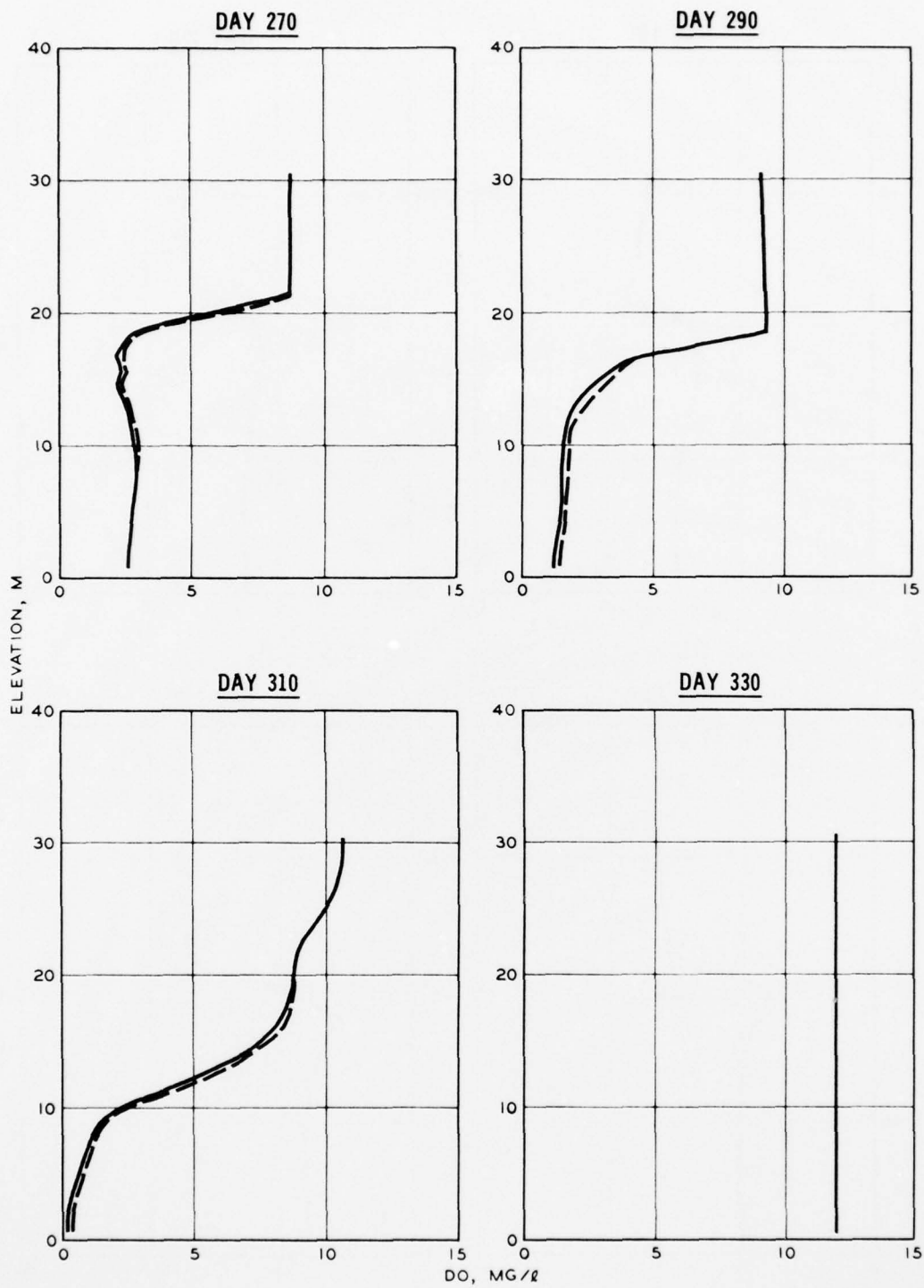


Figure 80 (sheet 3 of 3)

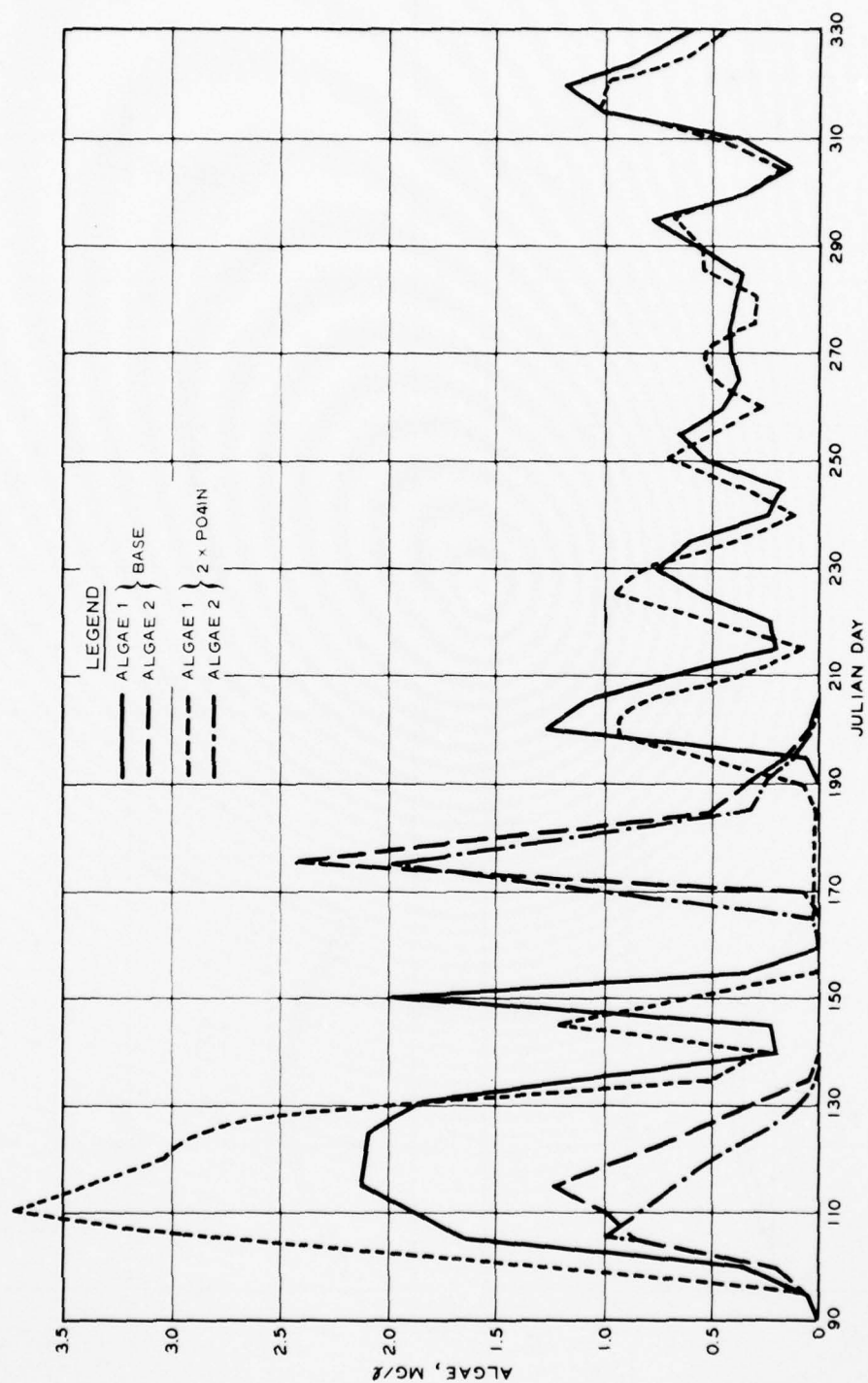


Figure 81. Effect of doubled phosphorus loadings on ALGAE 1 and 2 concentrations, 1973

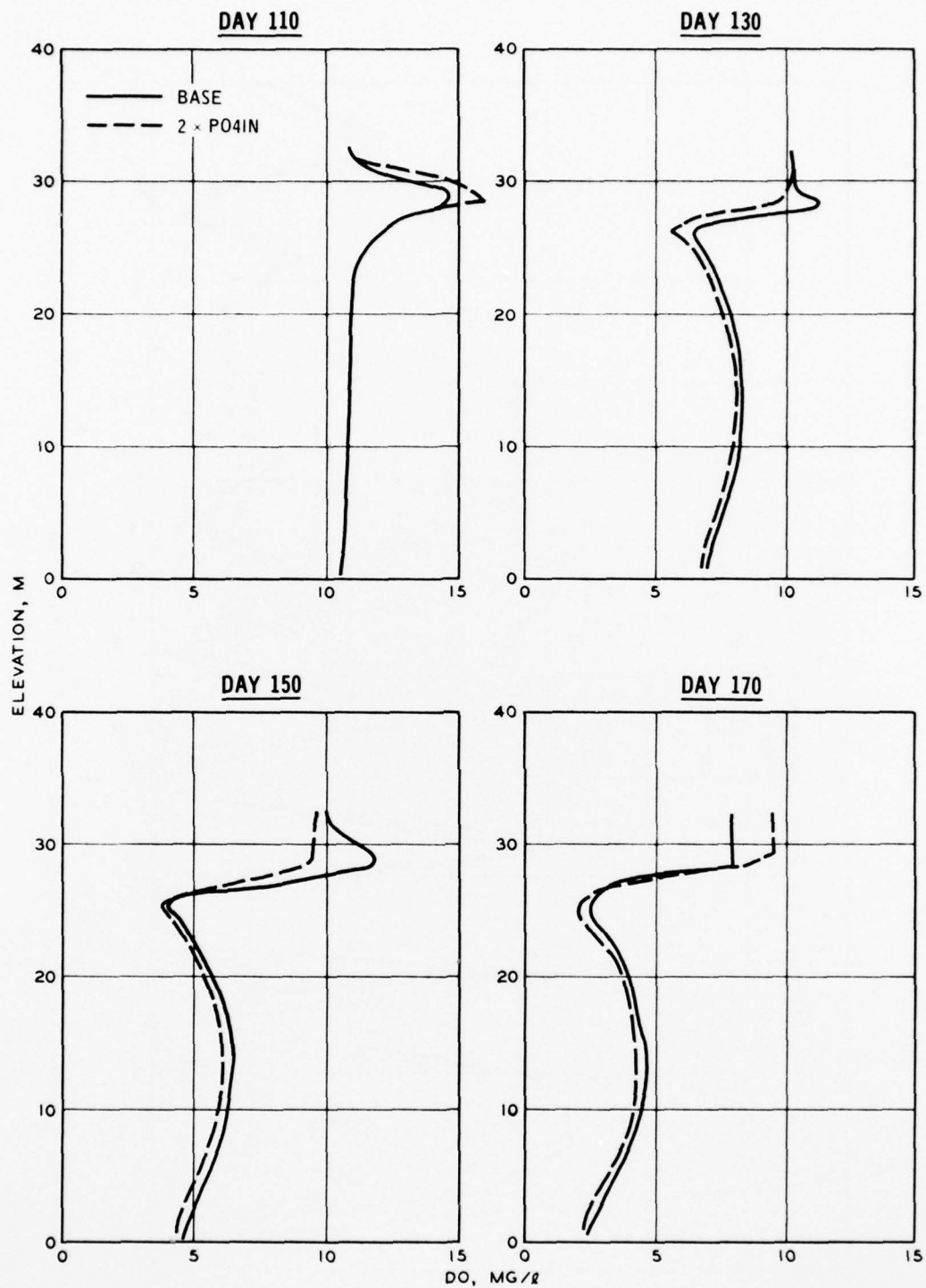


Figure 82. Effect of doubled phosphorus loadings on DO, 1973 (sheet 1 of 3)

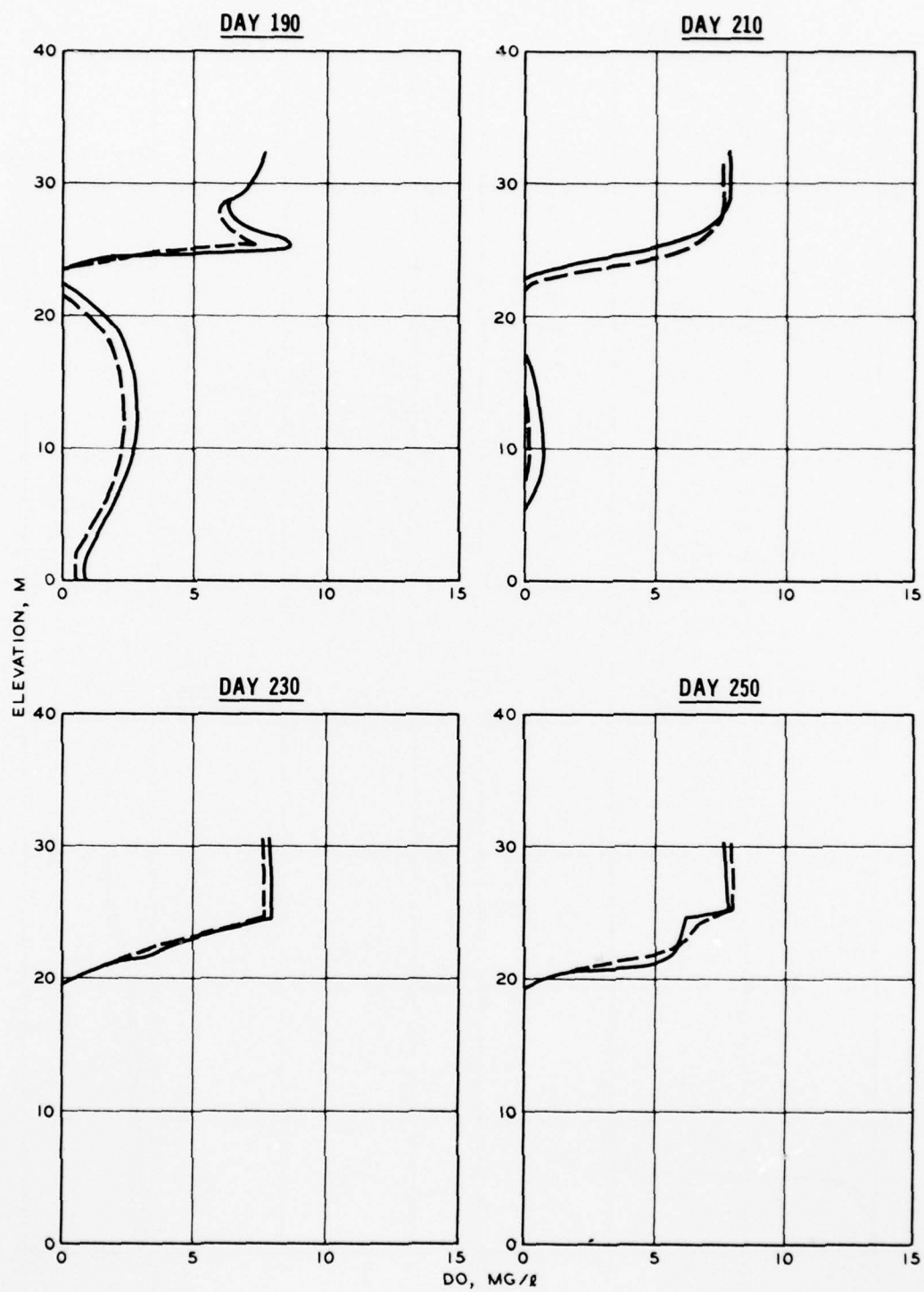


Figure 82 (sheet 2 of 3)

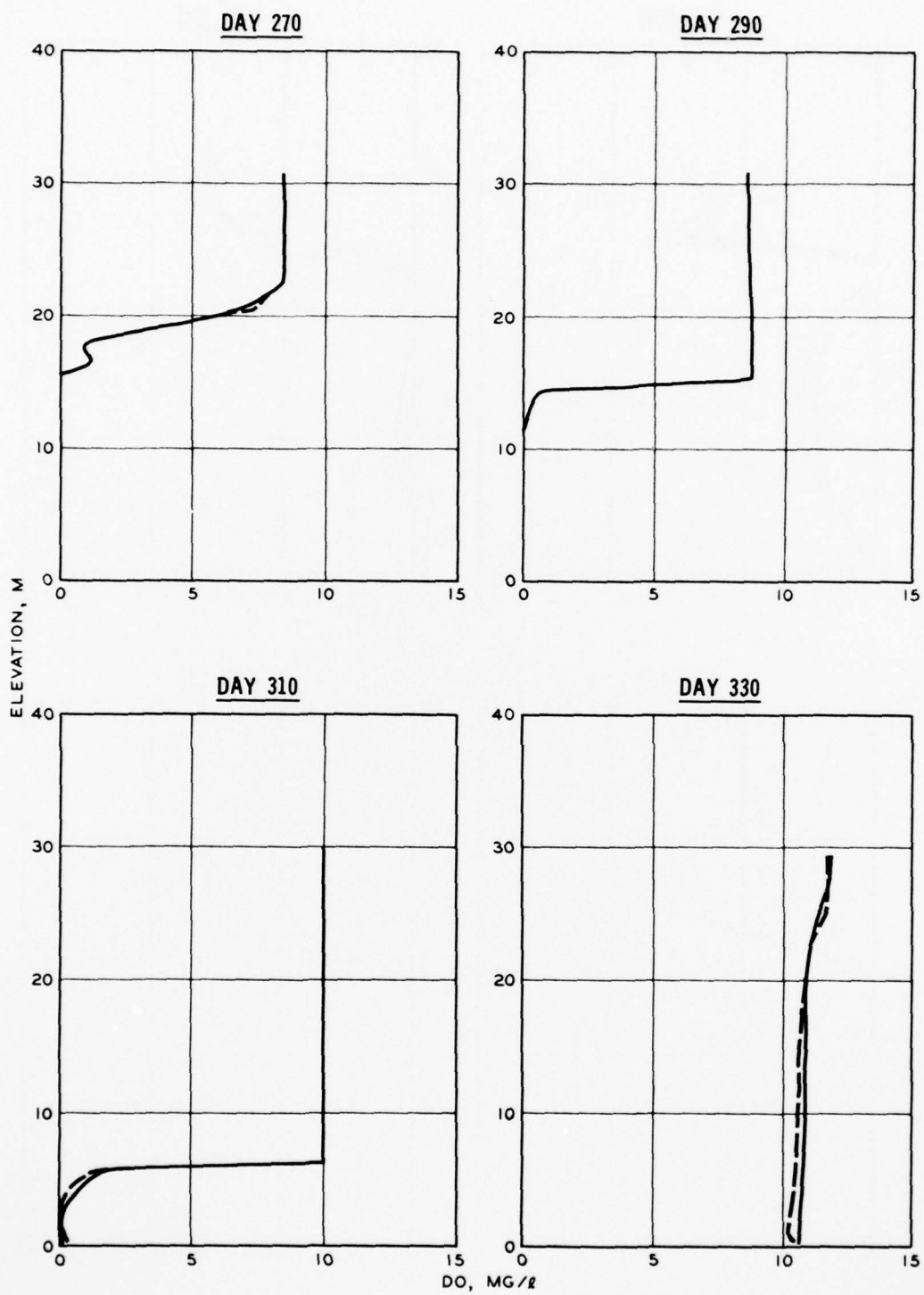


Figure 82 (sheet 3 of 3)

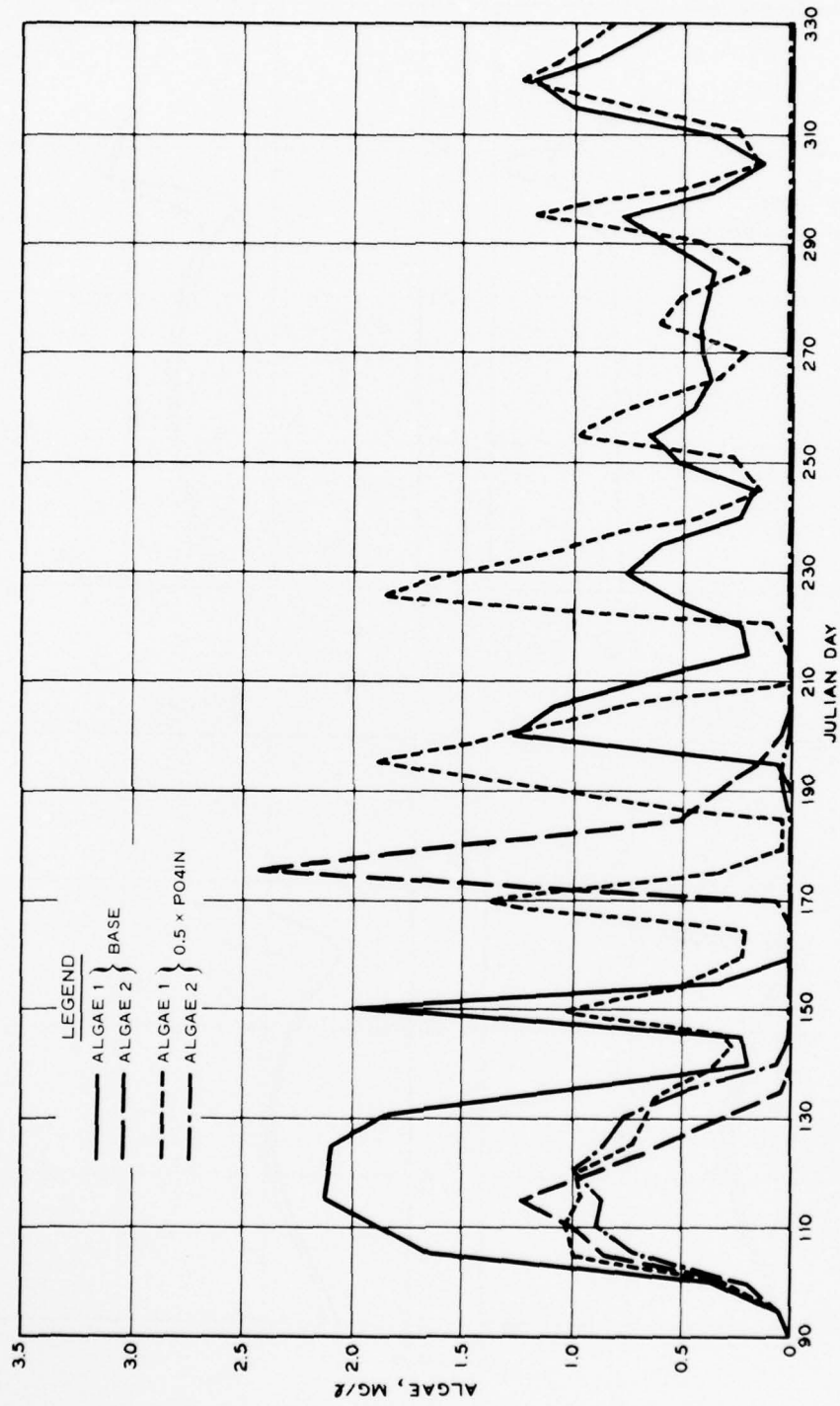


Figure 83. Effect of halved phosphorus loadings on ALGAE 1 and 2 concentrations, 1973

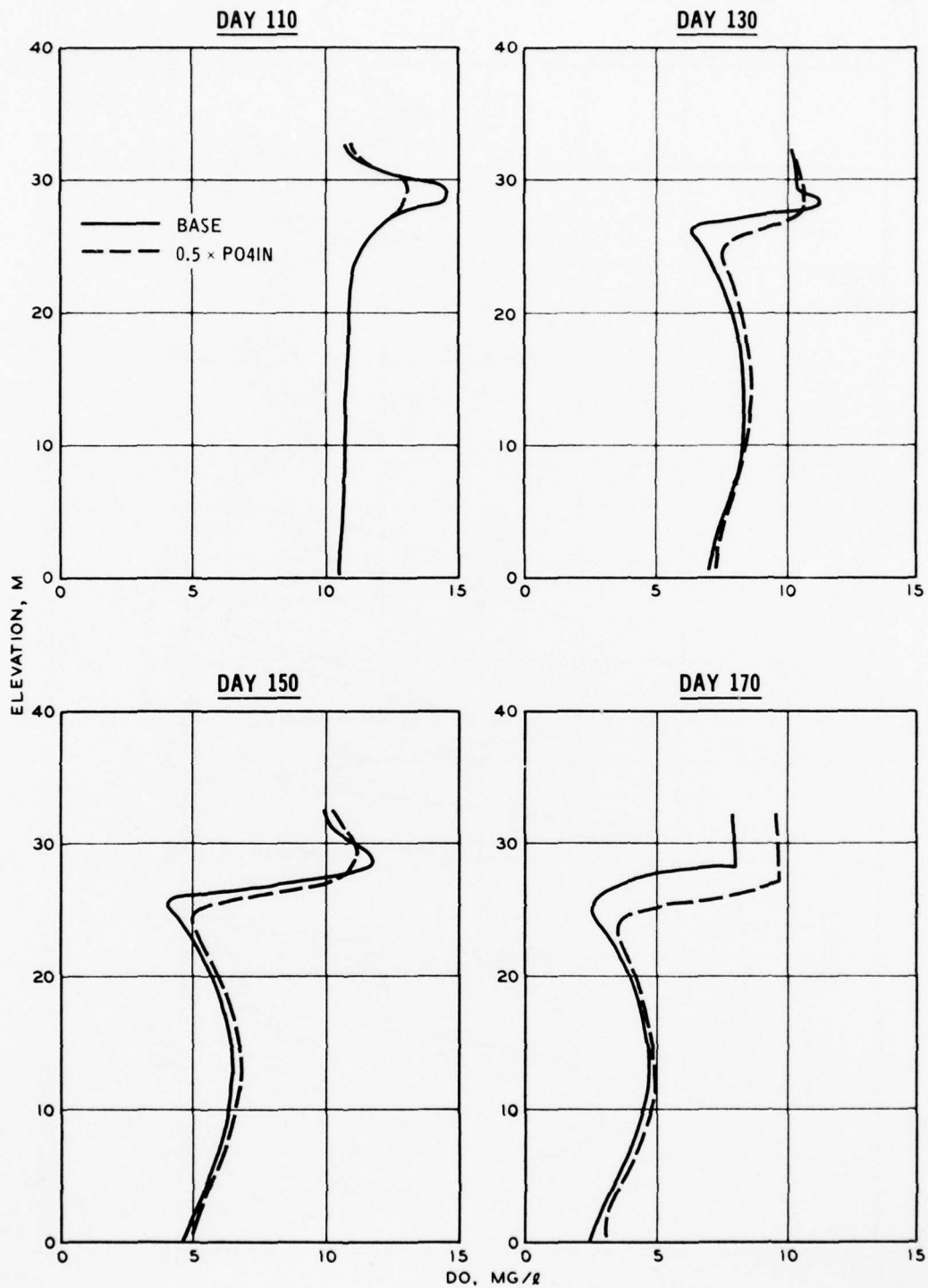


Figure 84. Effect of halved phosphorus loadings on DO, 1973 (sheet 1 of 3)

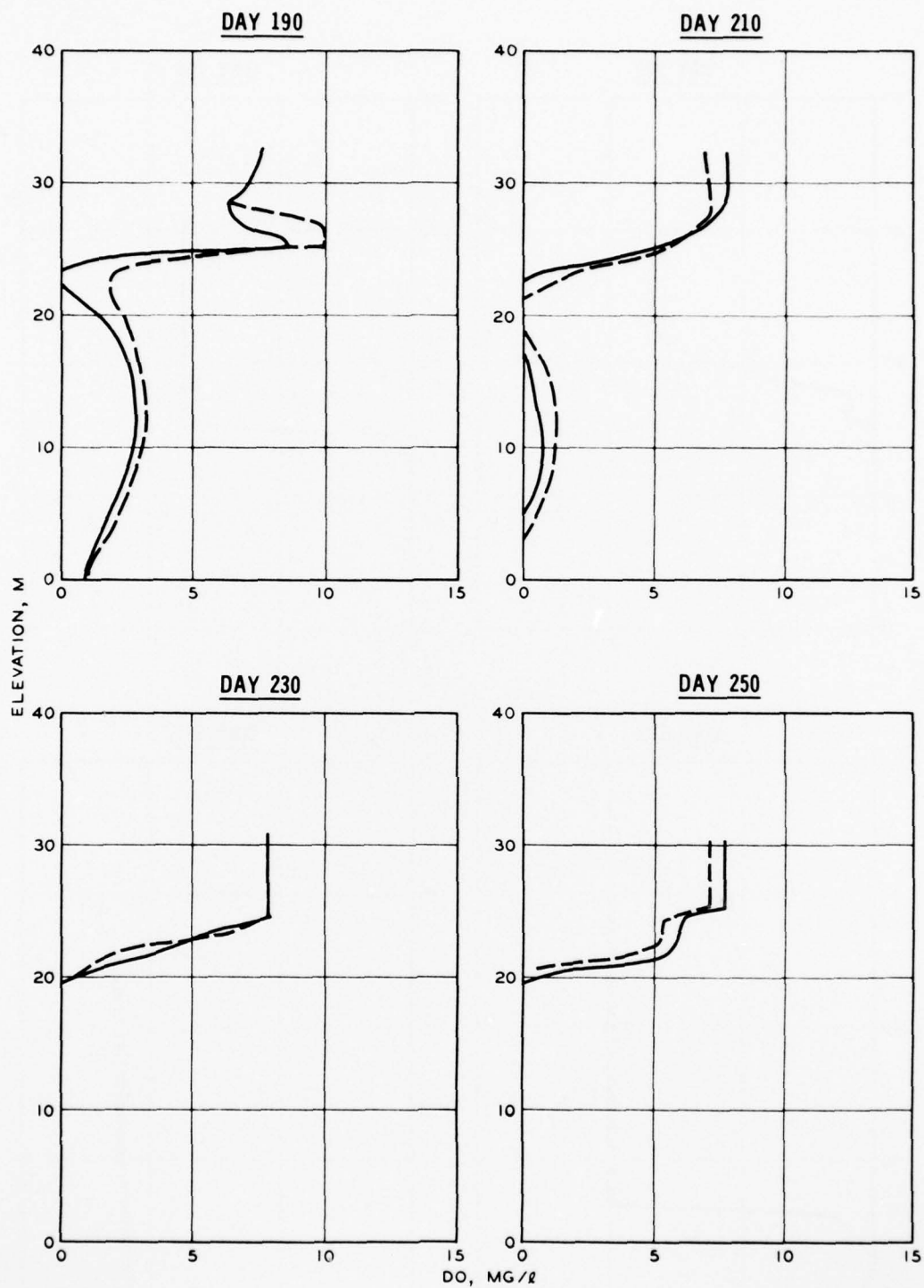


Figure 84 (sheet 2 of 3)

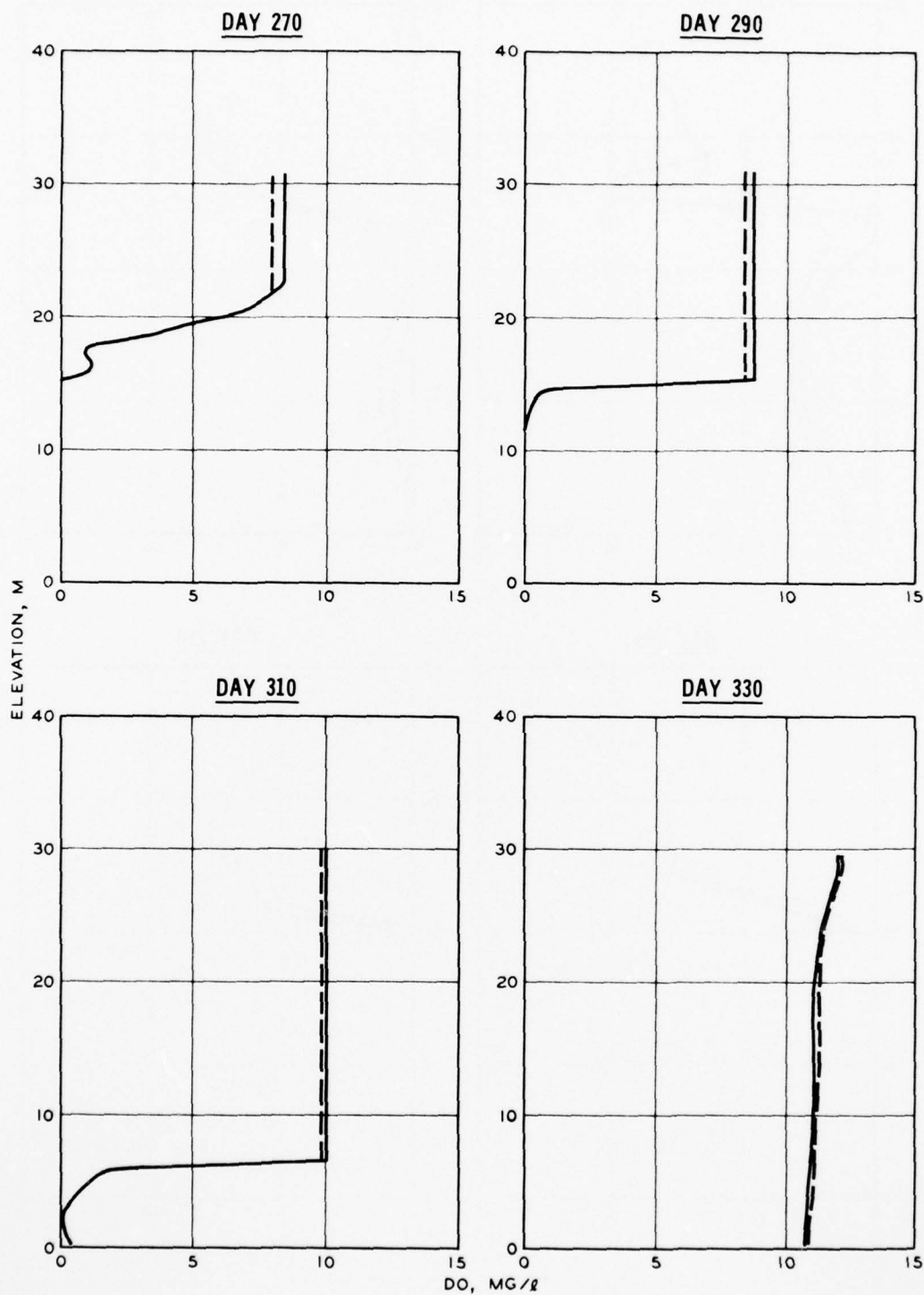


Figure 84 (sheet 3 of 3)

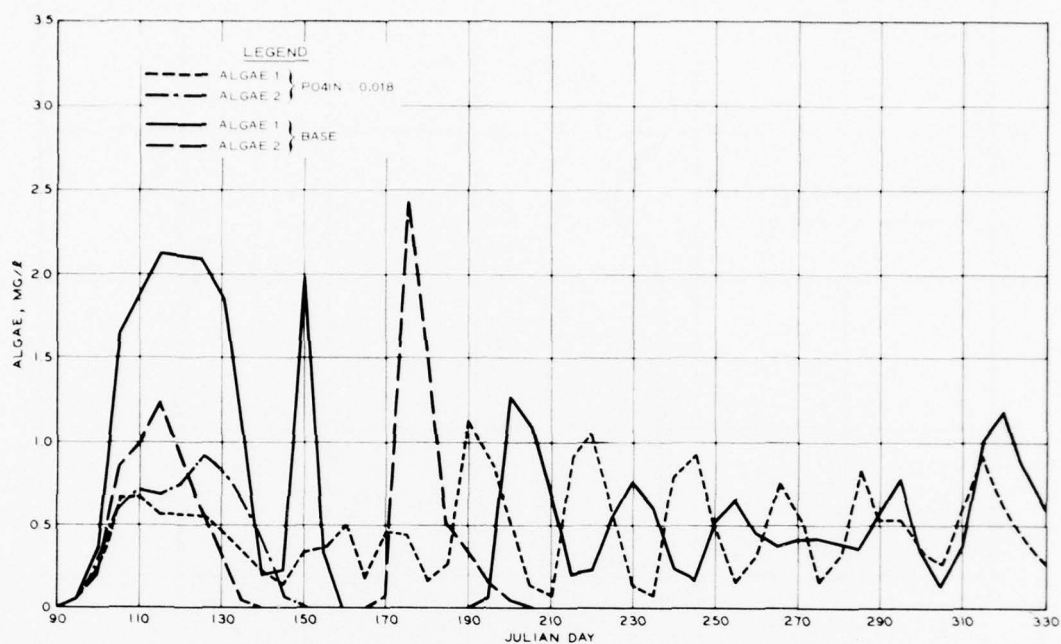


Figure 85. Comparison of algae concentrations for the mean phosphorus concentration and the base case, 1973

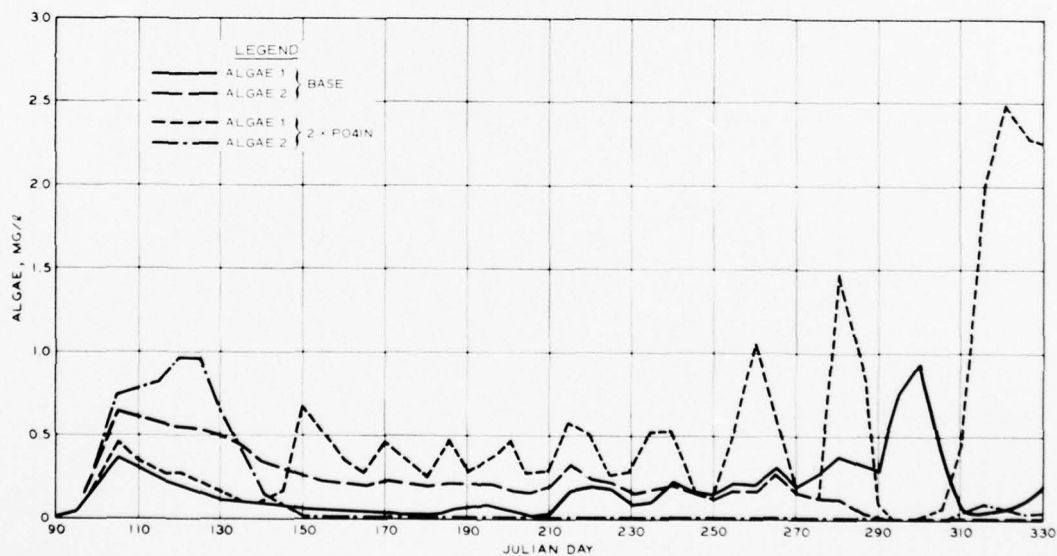


Figure 86. Effect of doubled phosphorus loadings on ALGAE 1 and 2 concentrations, 1969

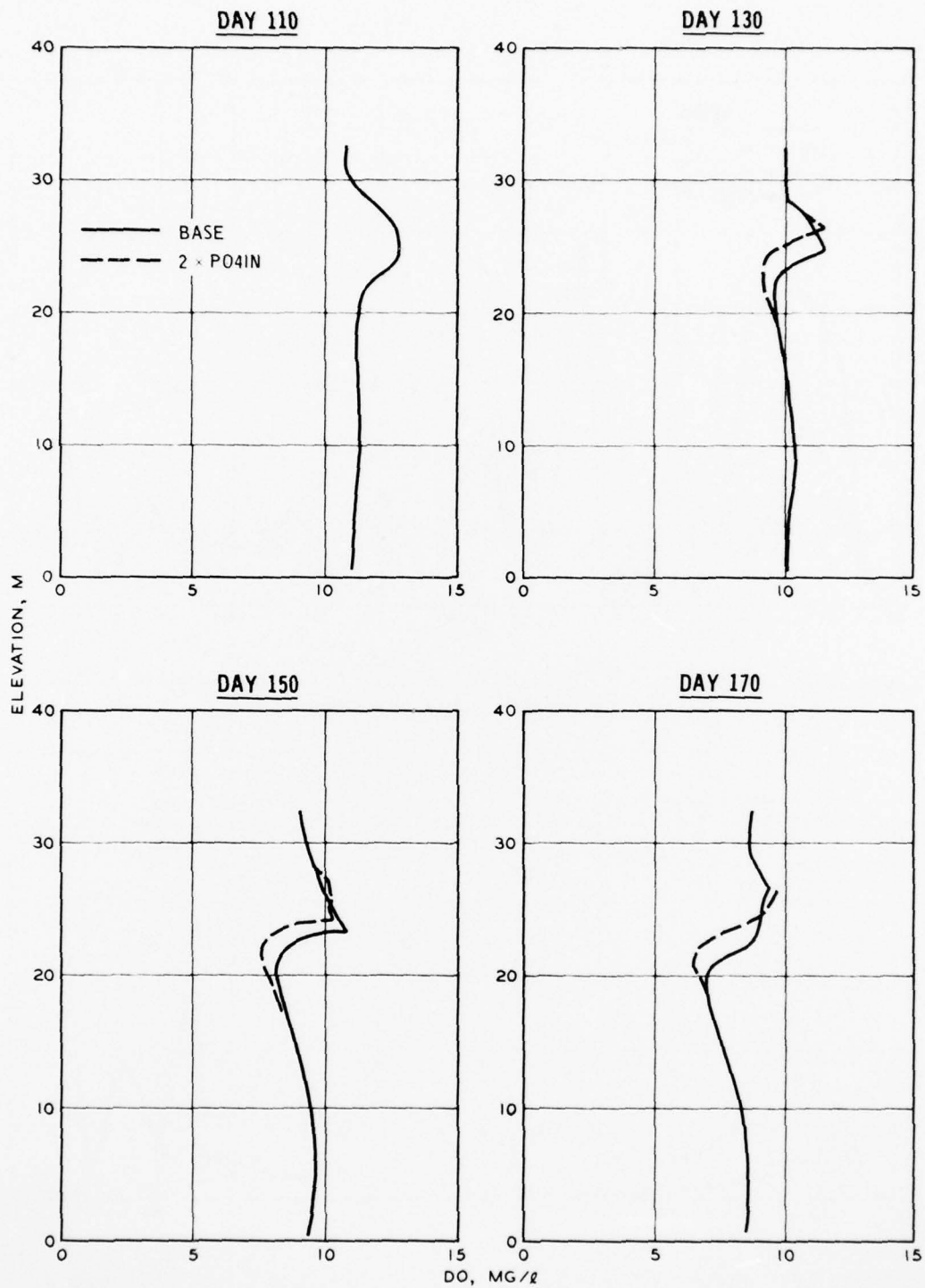


Figure 87. Effect of doubled phosphorus loadings on DO, 1969 (sheet 1 of 3)

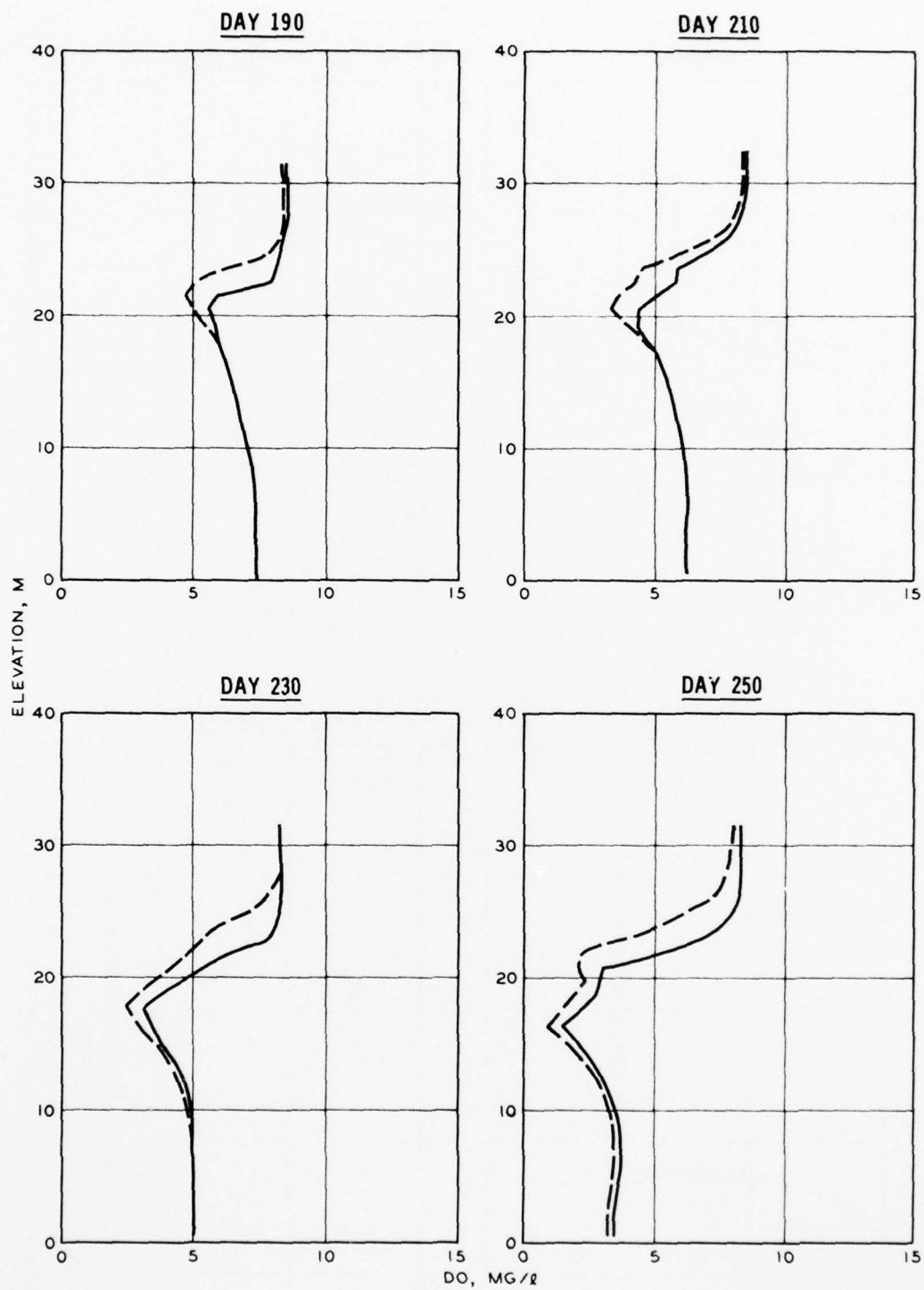


Figure 87 (sheet 2 of 3)

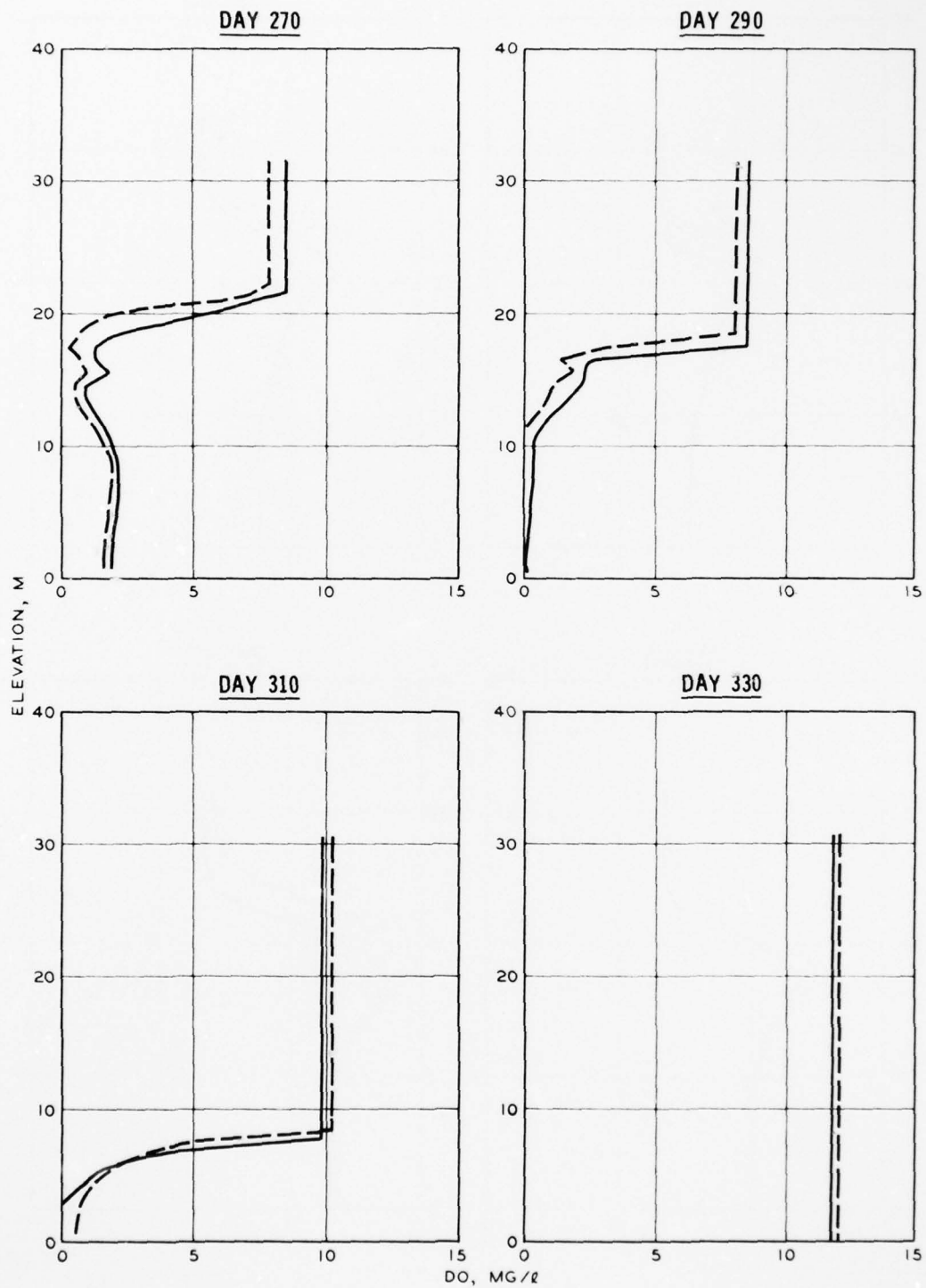


Figure 87 (sheet 3 of 3)

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WATER QUALITY EVALUATION OF PROPOSED TREXLER LAKE, JORDAN CREEK--ETC(U)
AUG 78 D E FORD, K W THORNTON, A S LESSEM

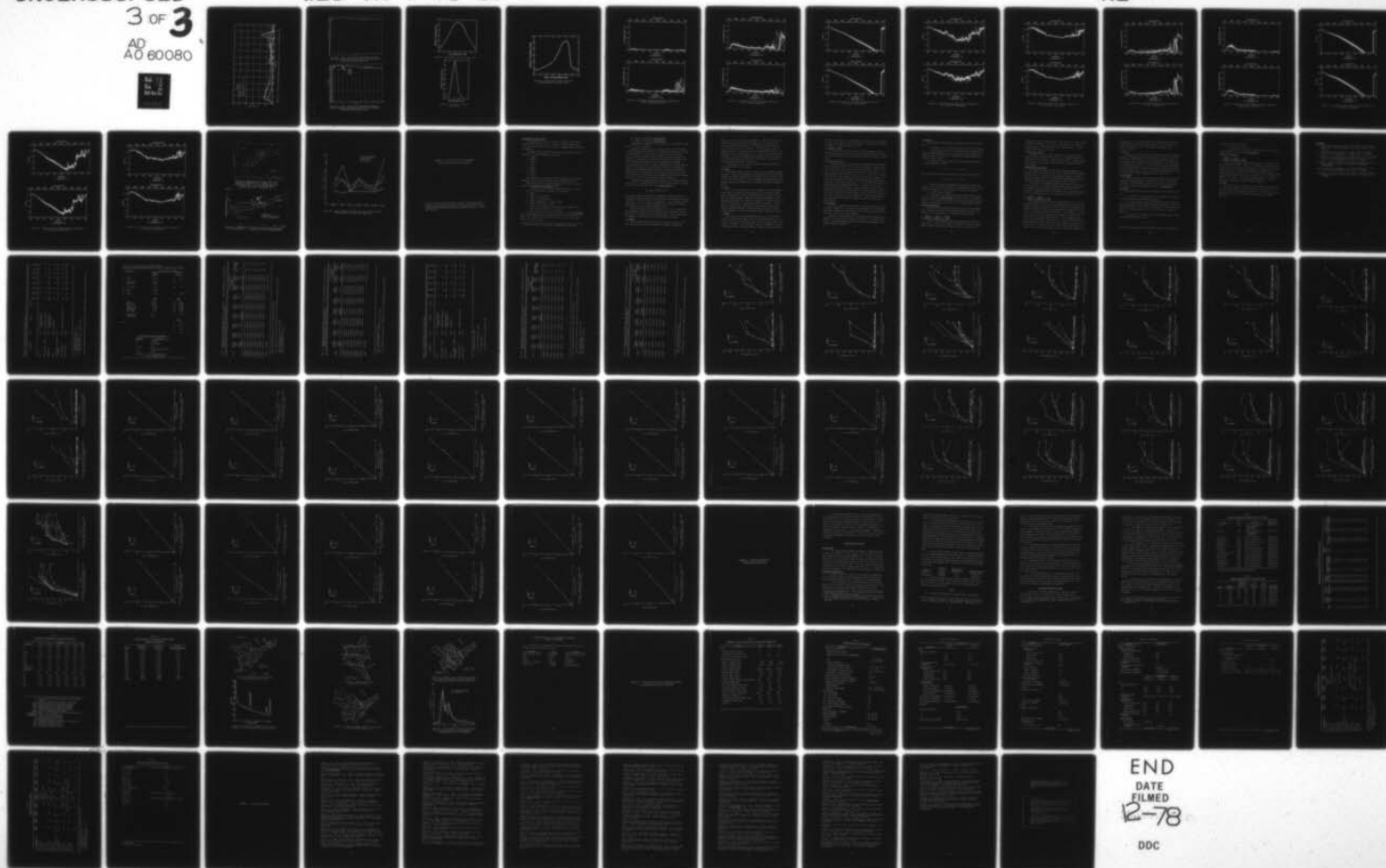
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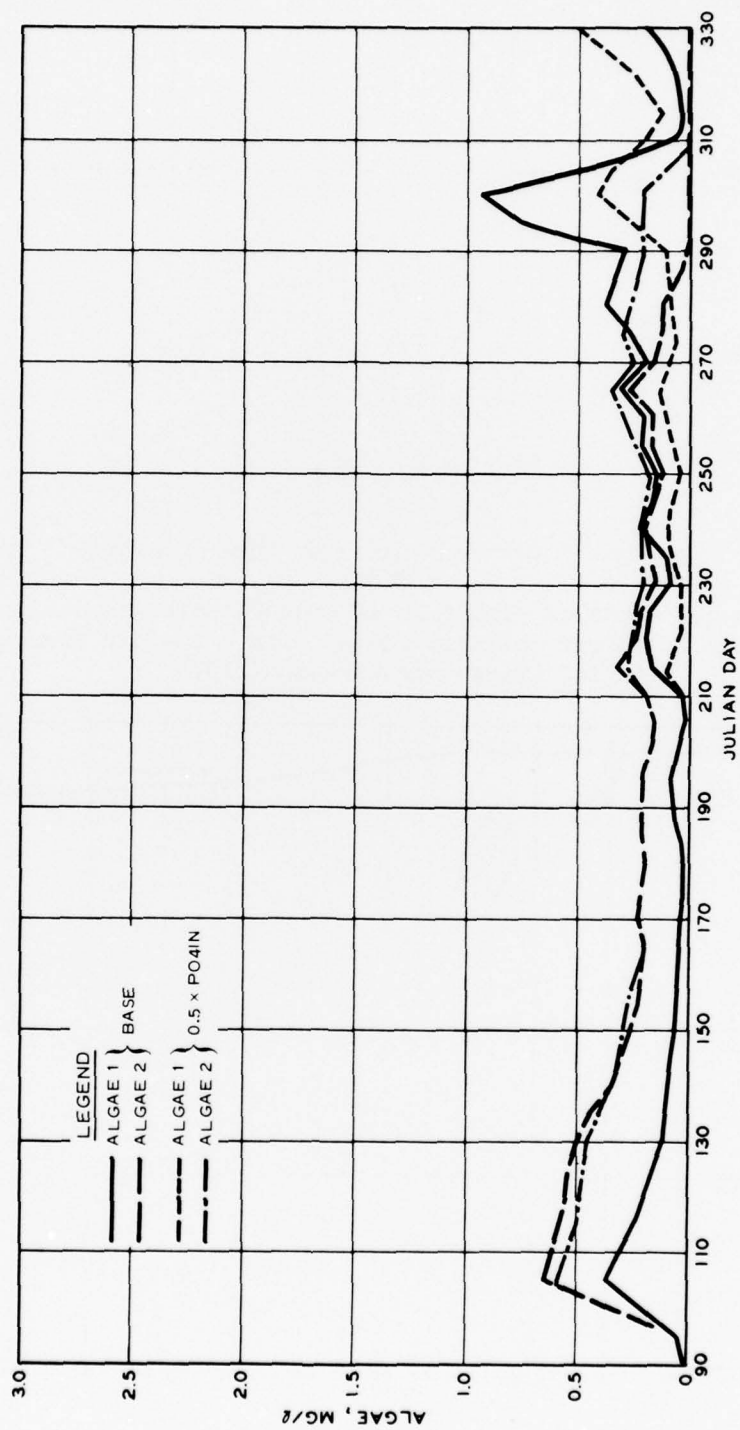


Figure 88. Effect of halved phosphorus loadings on ALGAE 1 and 2 concentrations, 1969

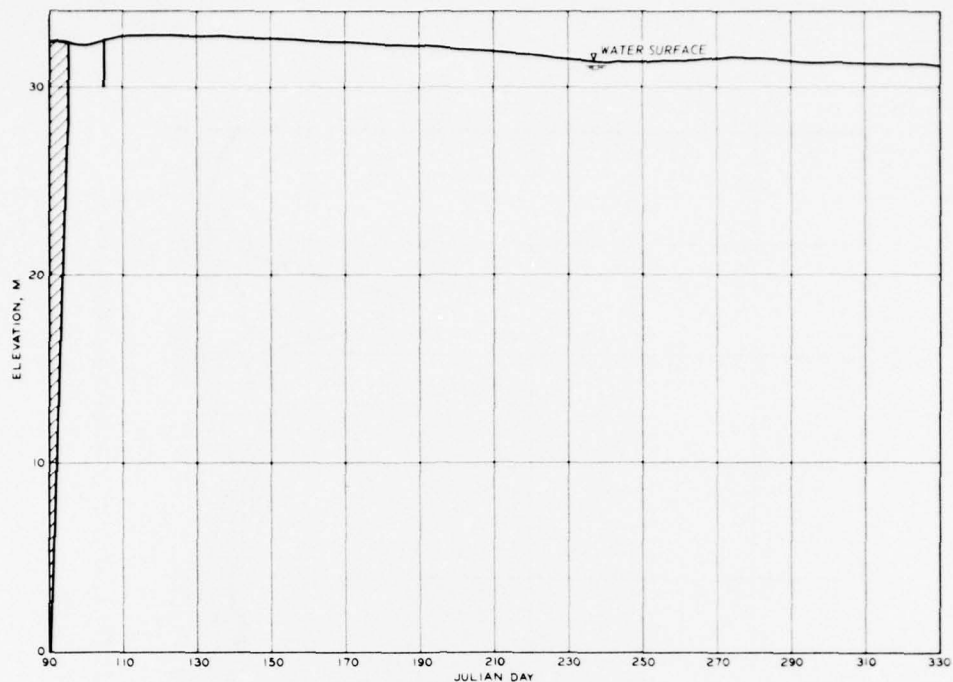


Figure 89. Zone of violation of Pennsylvania standard of 200 fecal coliform colonies/100 ml with a 10-fold increase in inflowing concentration, 1974

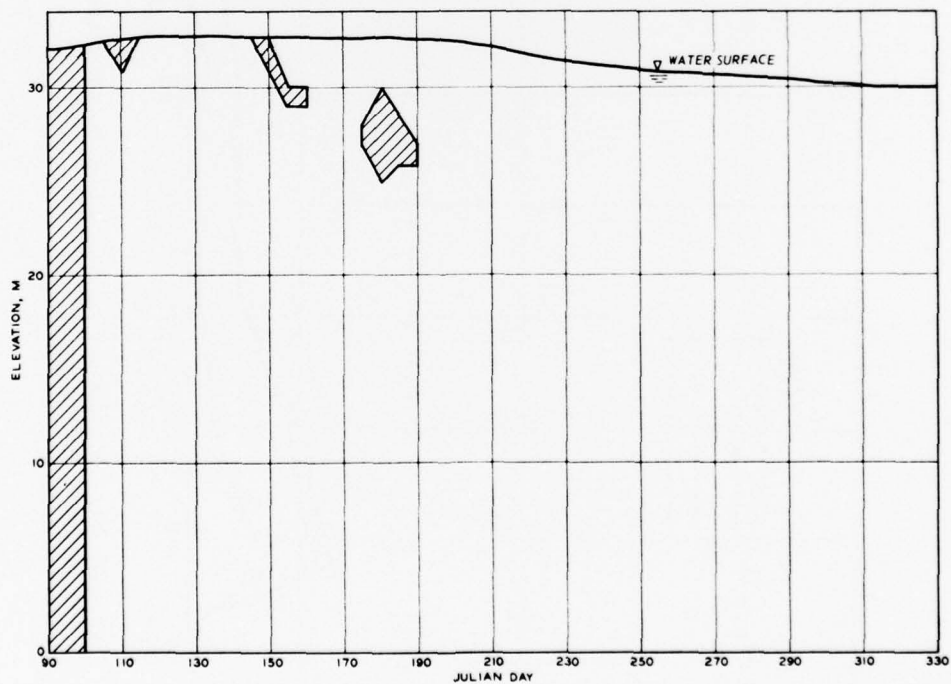


Figure 90. Zone of violation of Pennsylvania standard of 200 fecal coliform colonies/100 ml with a 10-fold increase in inflowing concentration, 1973

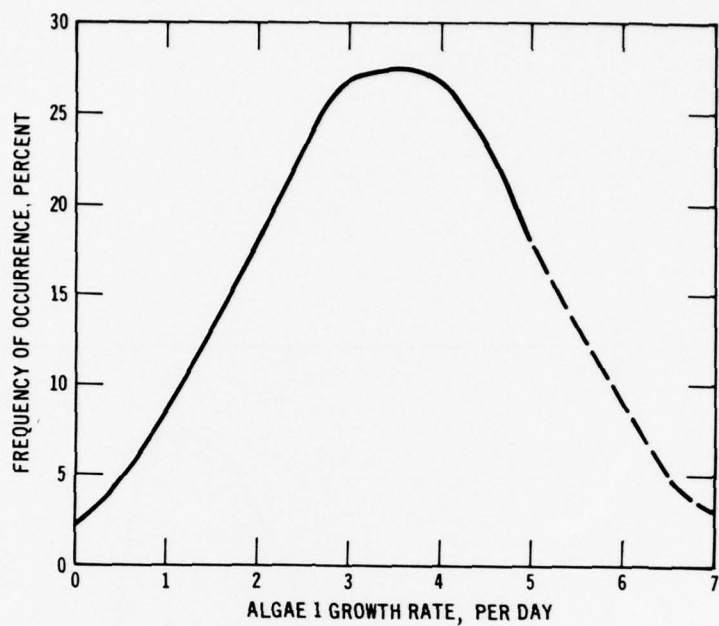


Figure 91. Distribution of ALGAE 1 growth rates

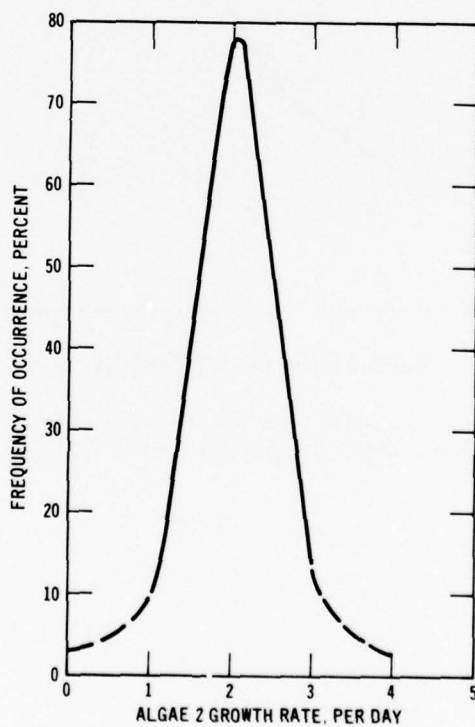


Figure 92. Distribution of ALGAE 2 growth rates

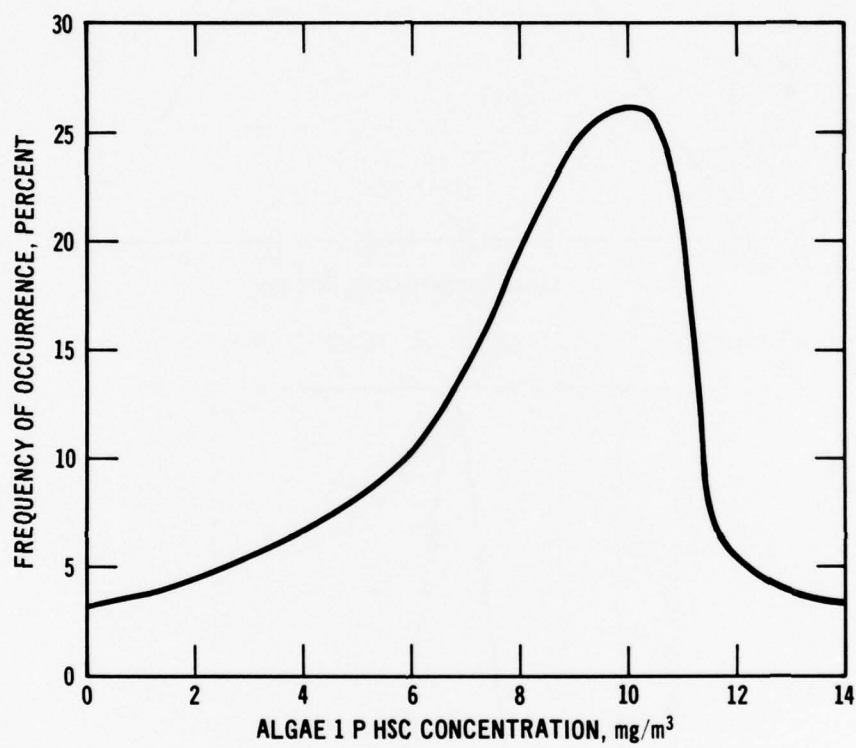


Figure 93. Distribution of ALGAE 1 phosphorus half-saturation coefficients (HSC)

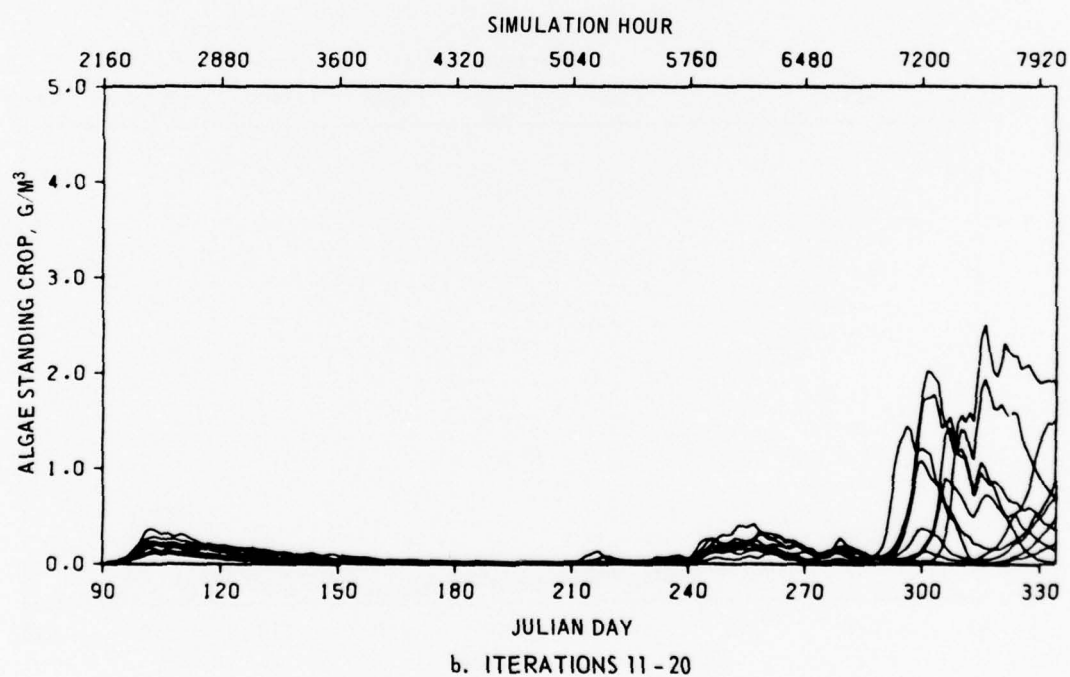
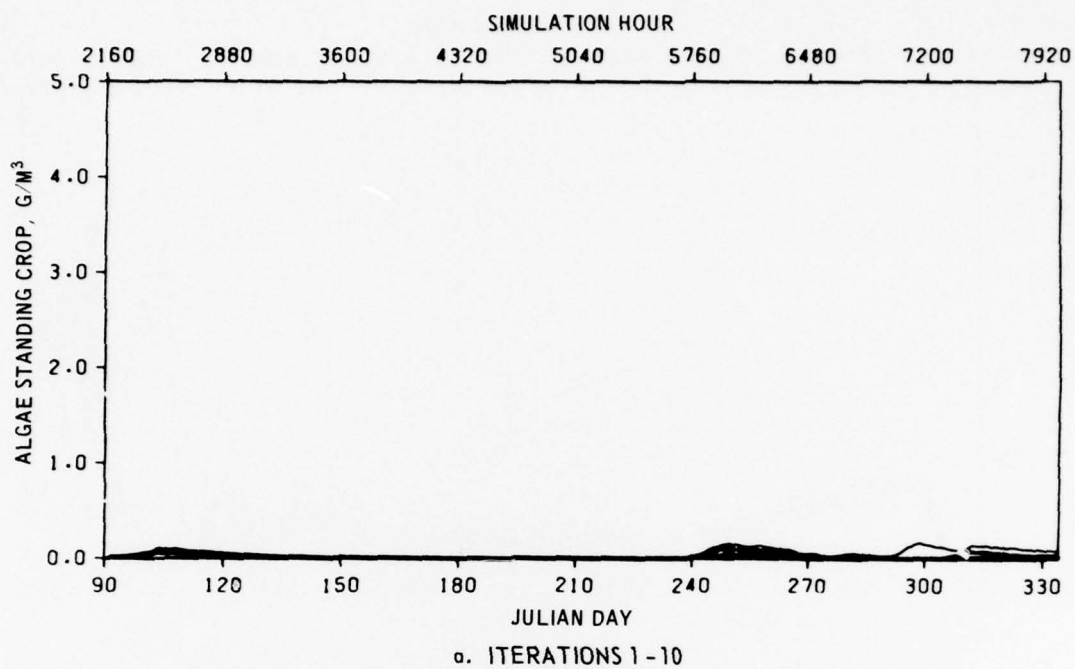


Figure 94. Monte Carlo coefficient variation simulations of
ALGAE 1 average standing crop

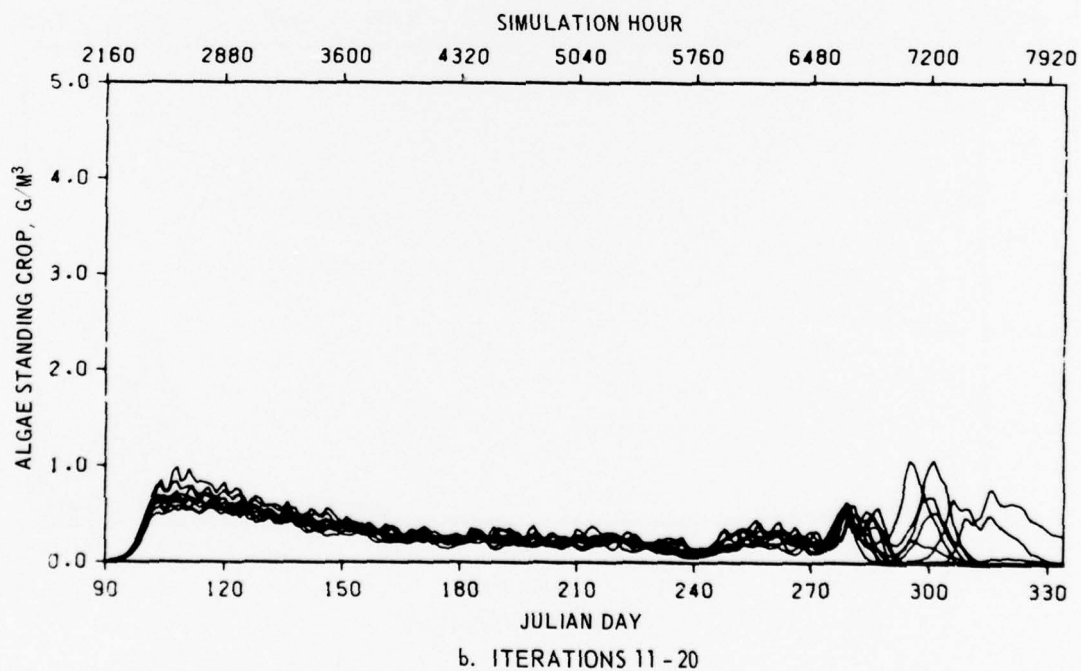
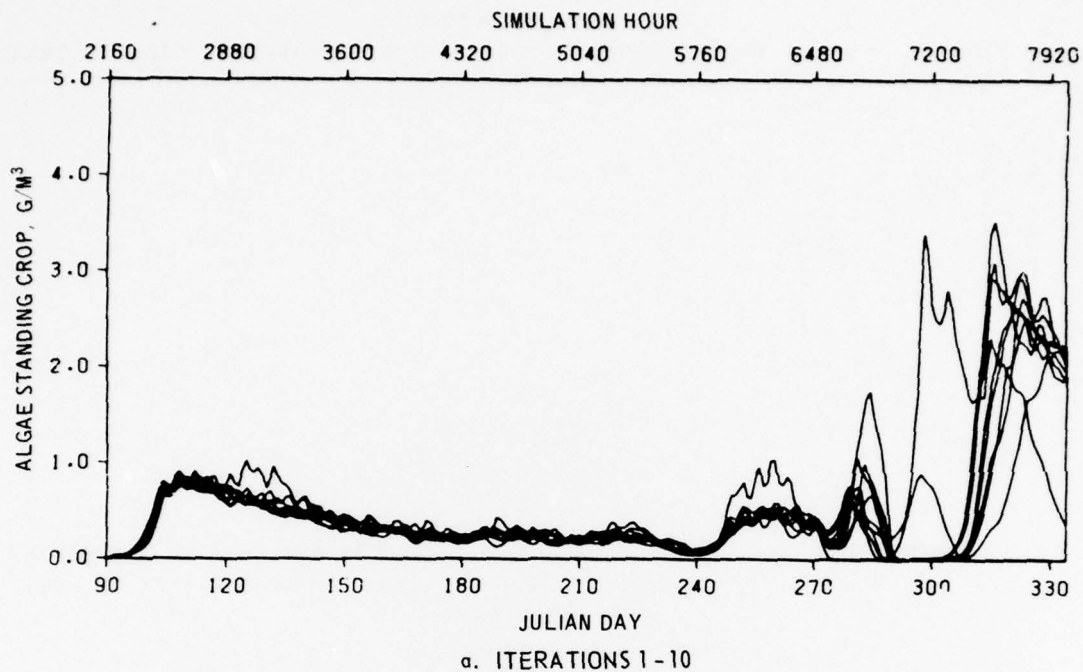


Figure 95. Monte Carlo coefficient variation simulation of
 ALGAE 2 average standing crop

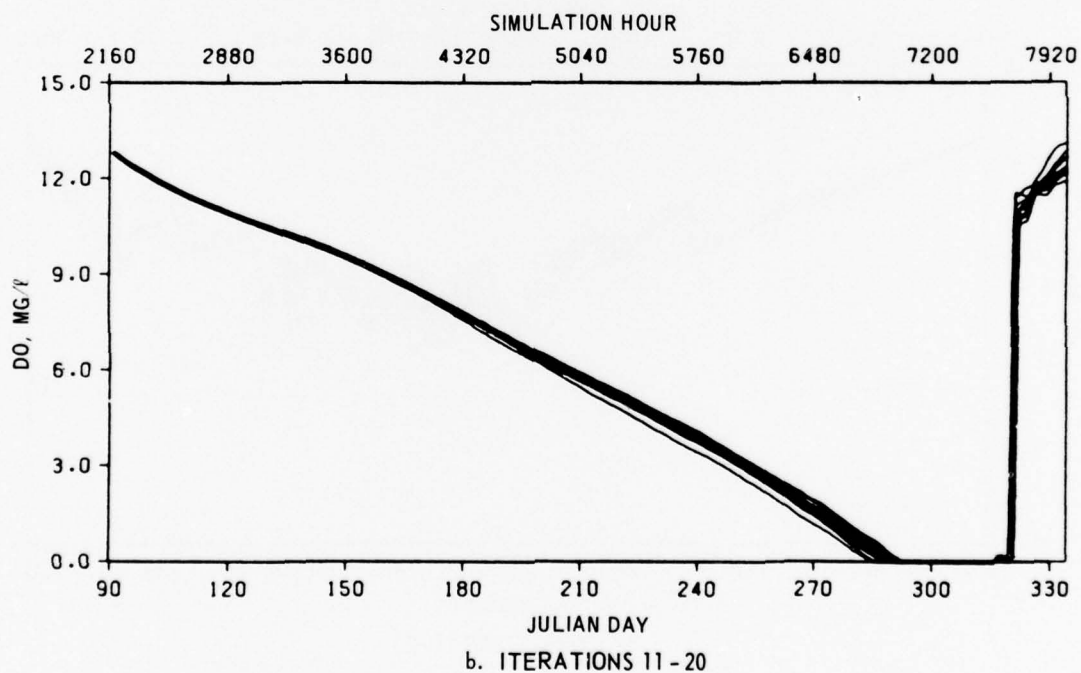
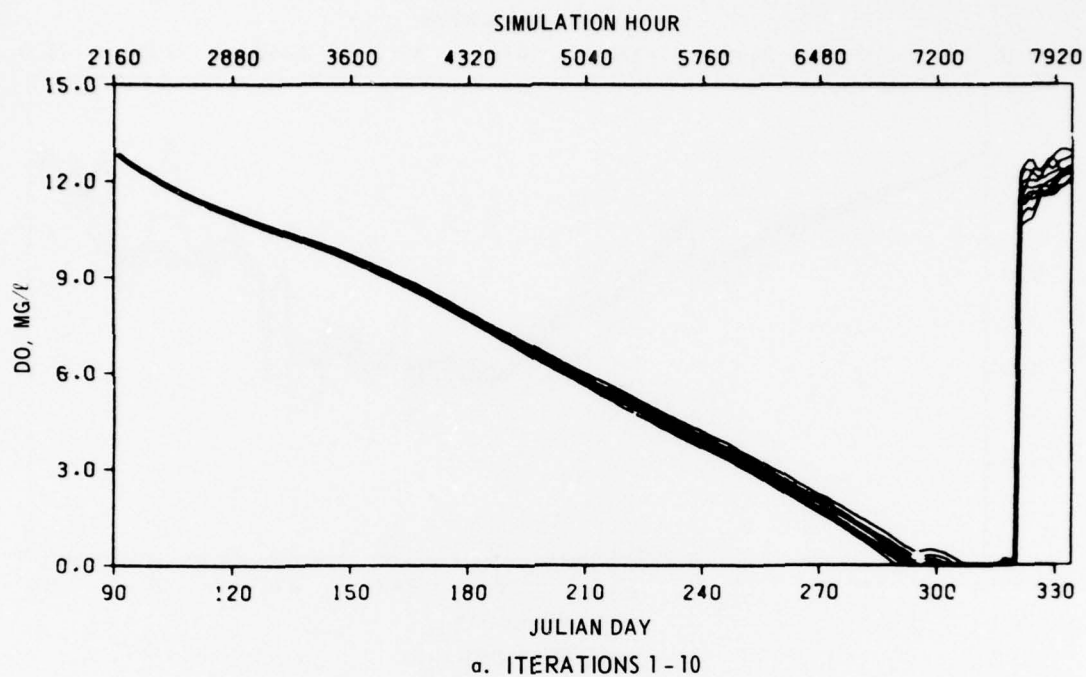


Figure 96. Monte Carlo coefficient variation simulation of DO in layer 2 (hypolimnion)

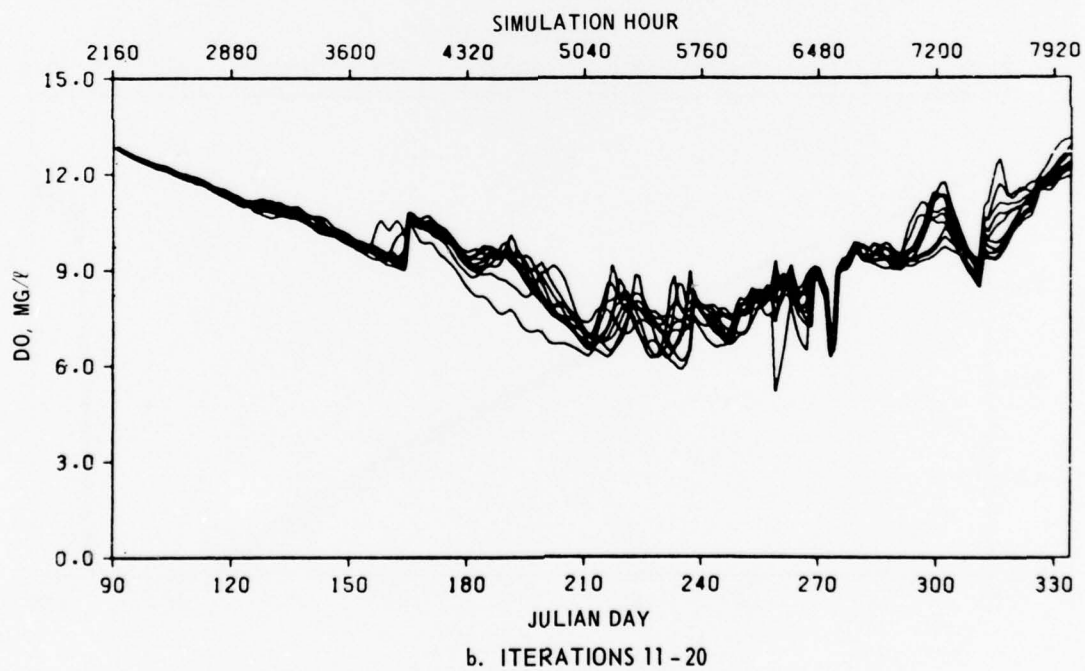
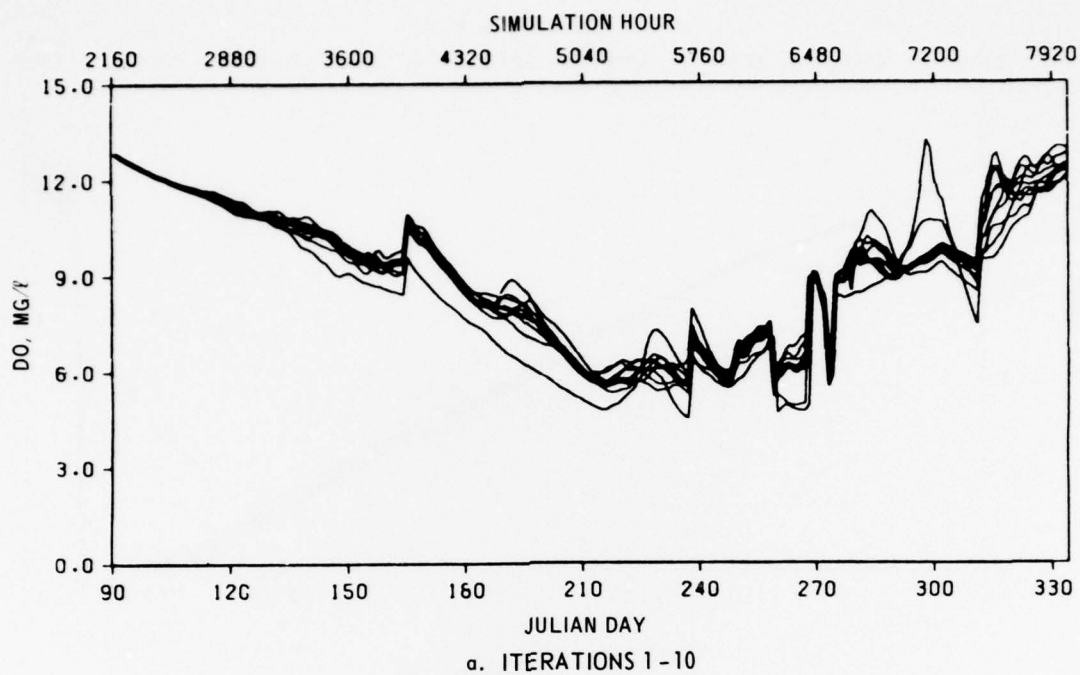


Figure 97. Monte Carlo coefficient variation simulation of
 DO in layer 22 (metalimnion)

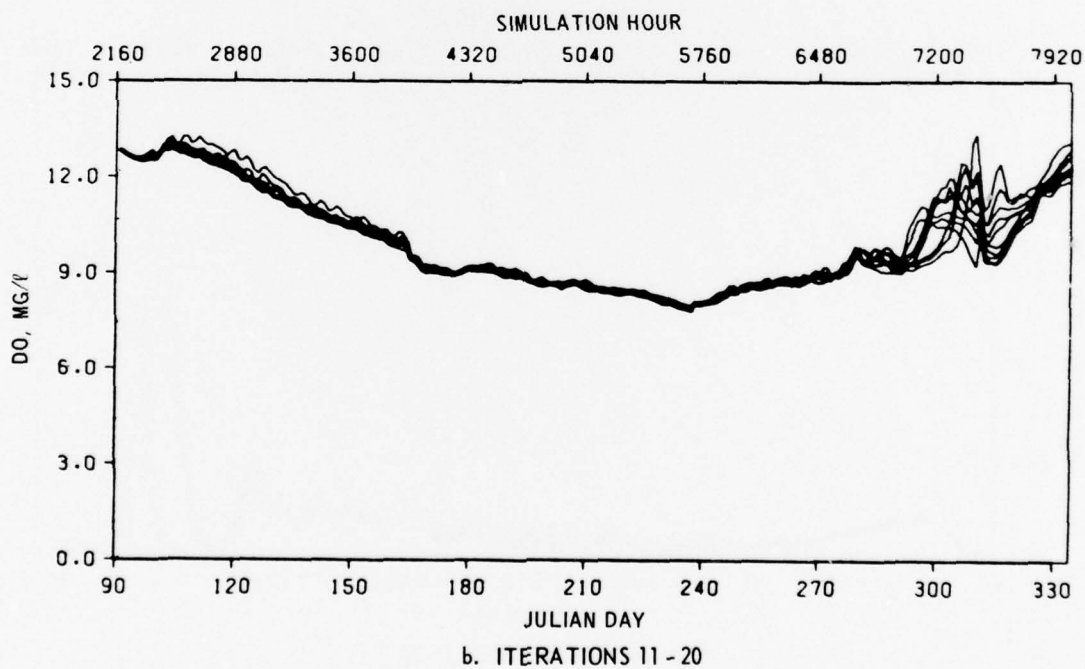
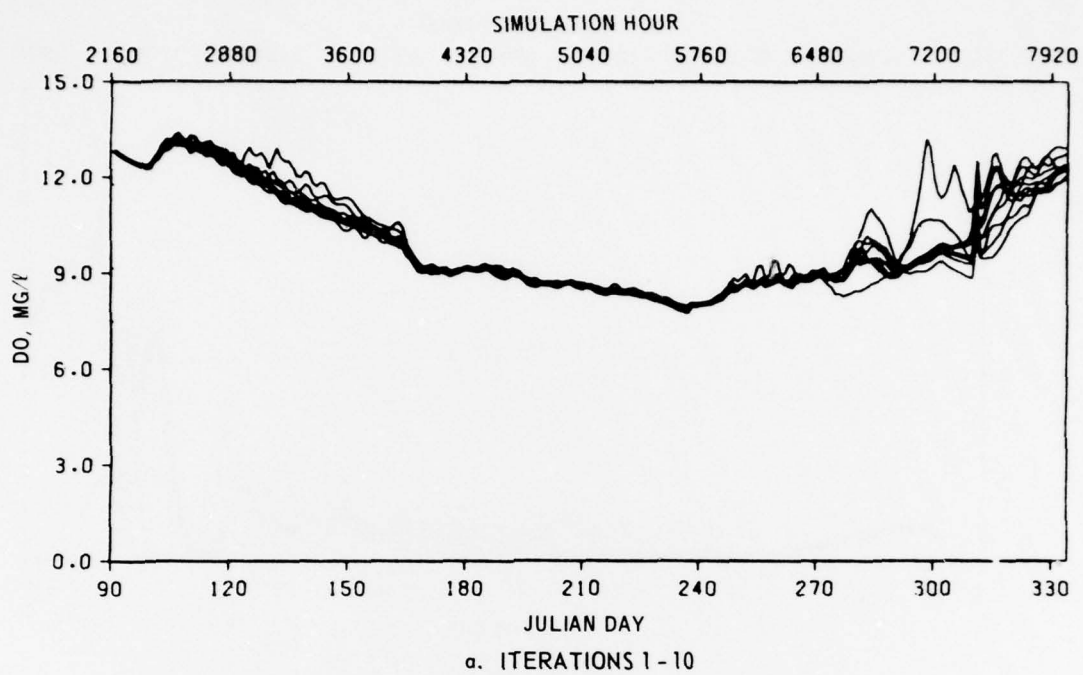
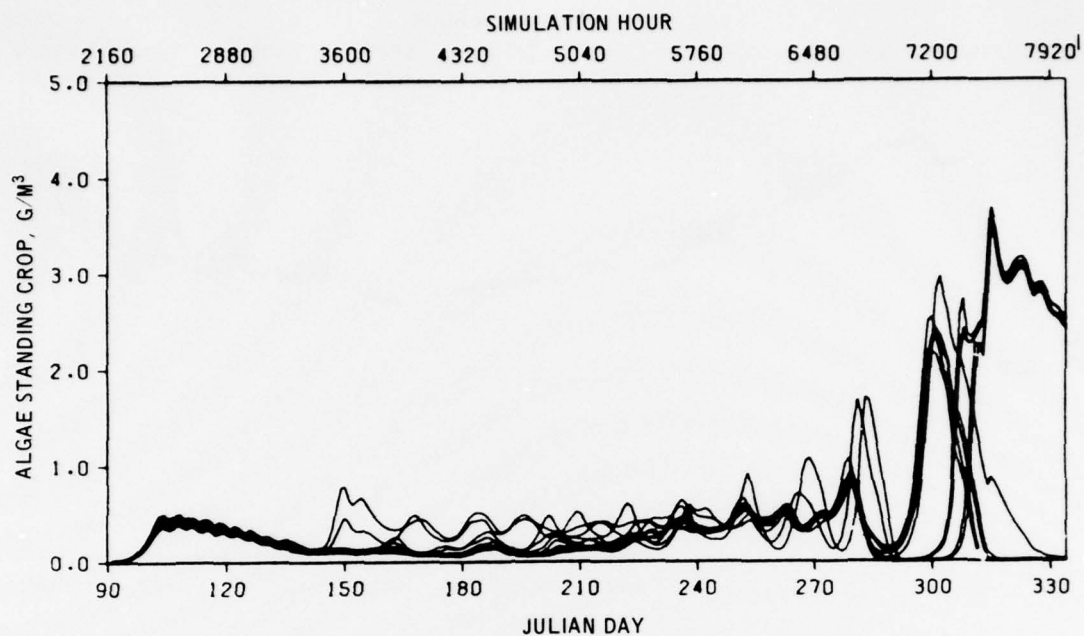
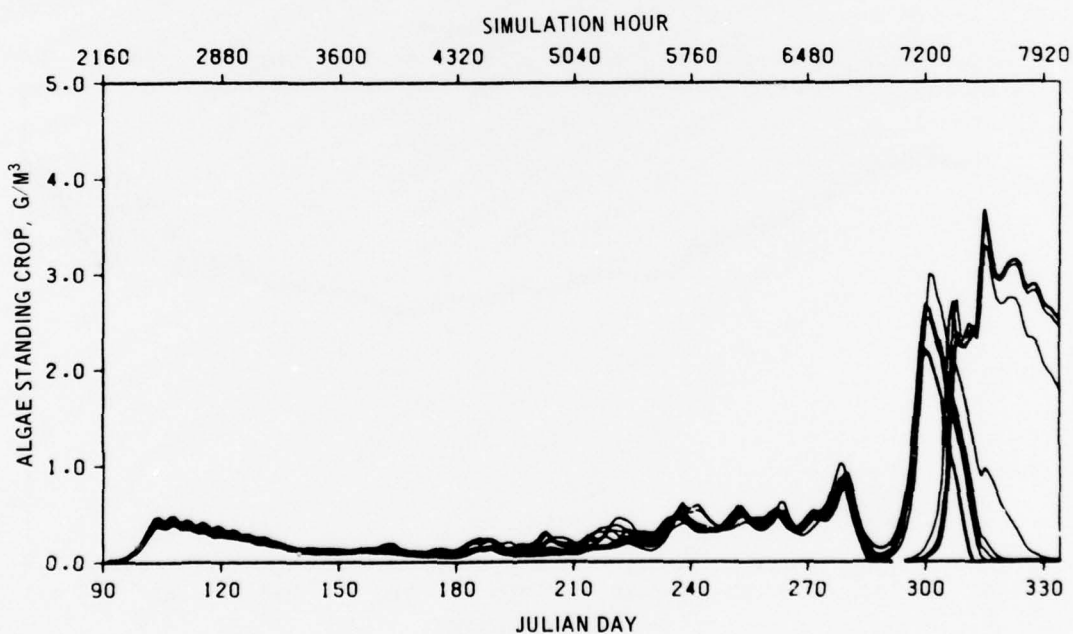


Figure 98. Monte Carlo coefficient variation simulation of DO in layer 27 (epilimnion)



a. ITERATIONS 1-10



b. ITERATIONS 11-20

Figure 99. Monte Carlo uniform update variation simulation of ALGAE 1 standing crop

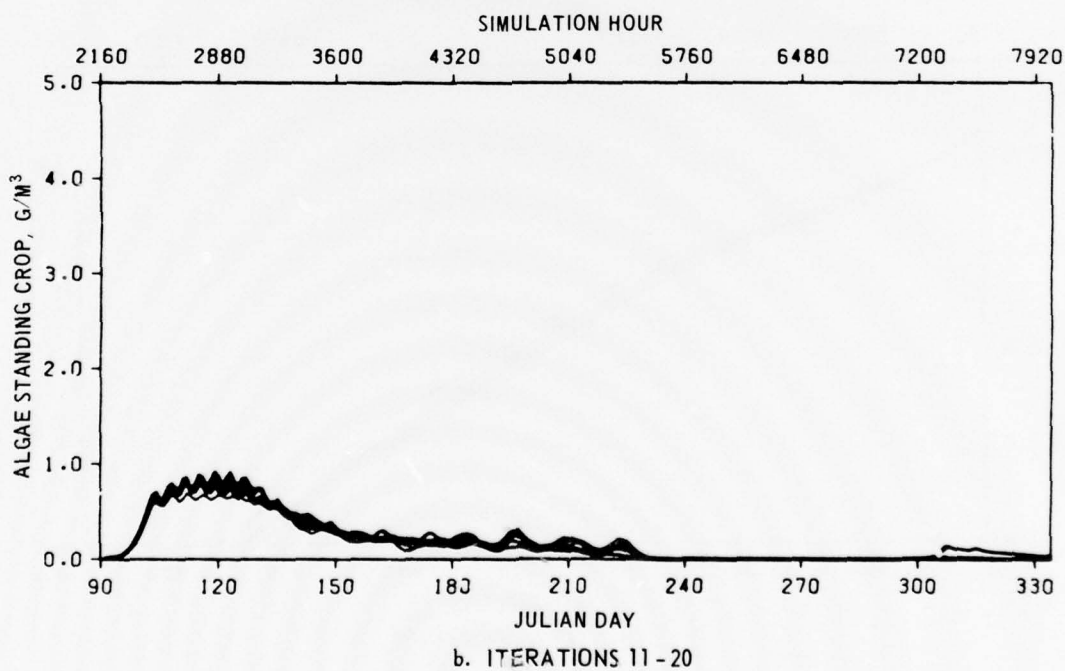
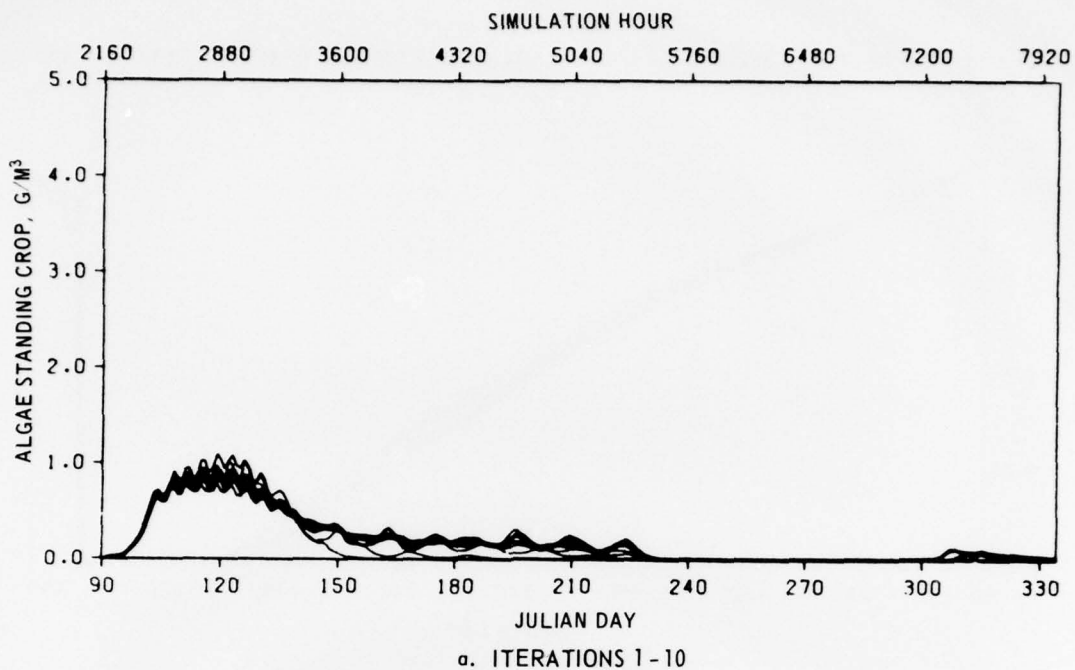


Figure 100. Monte Carlo uniform update variation simulation of average ALGAE 2 standing crop

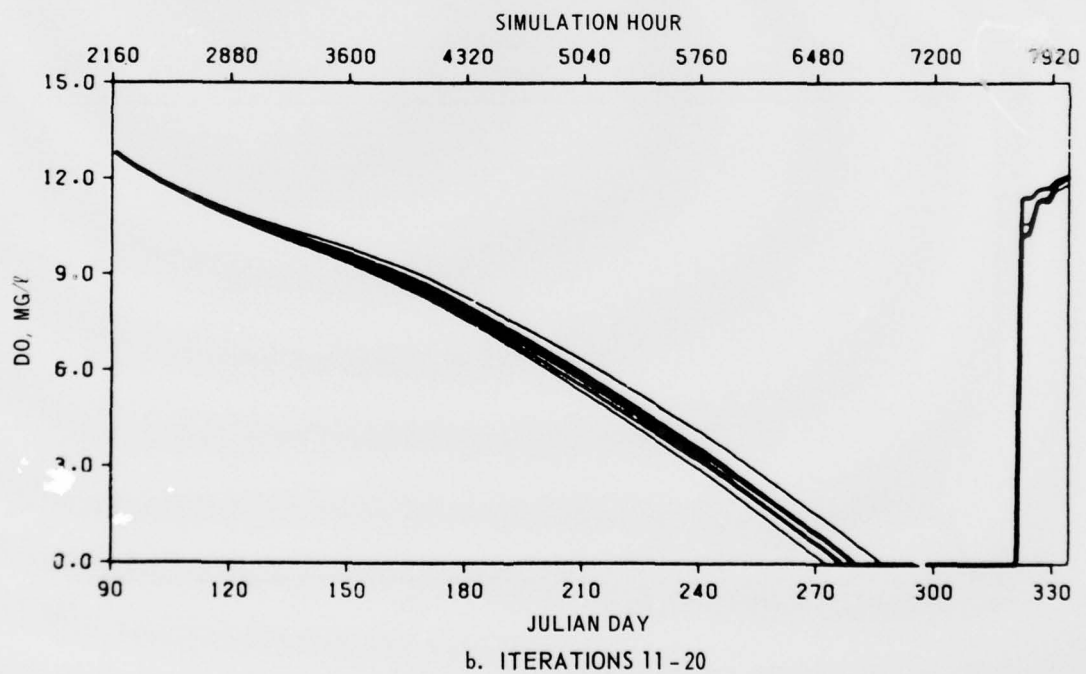
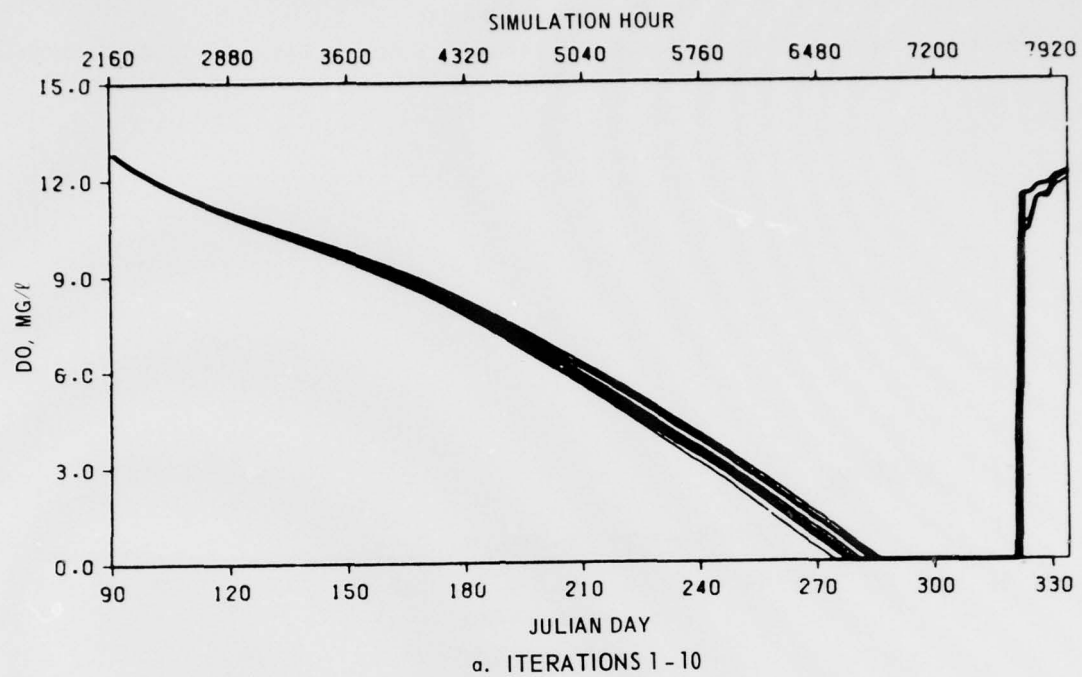
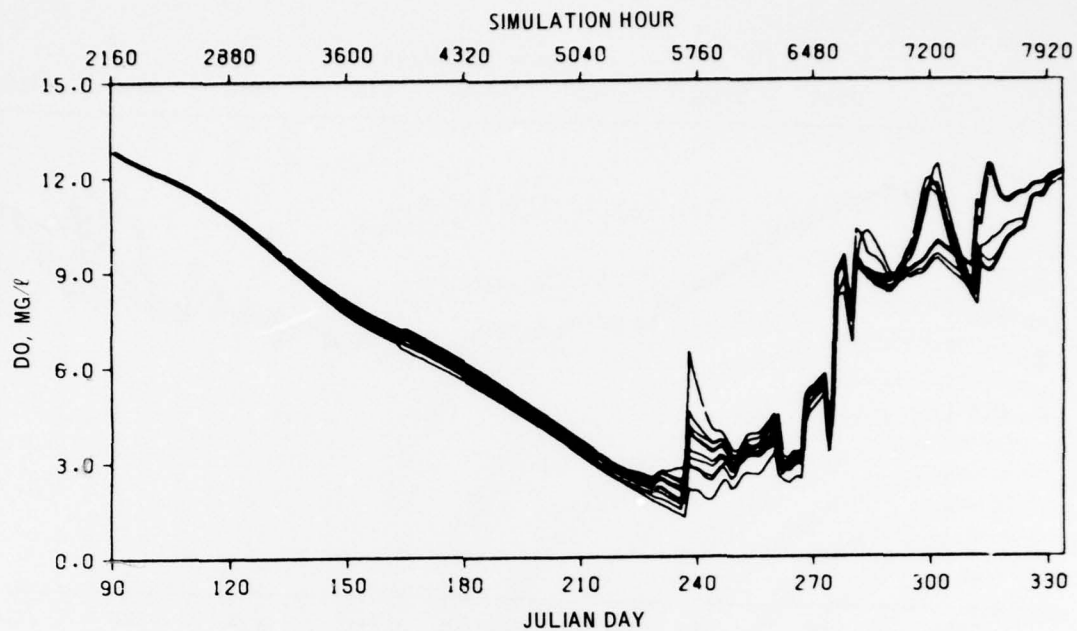
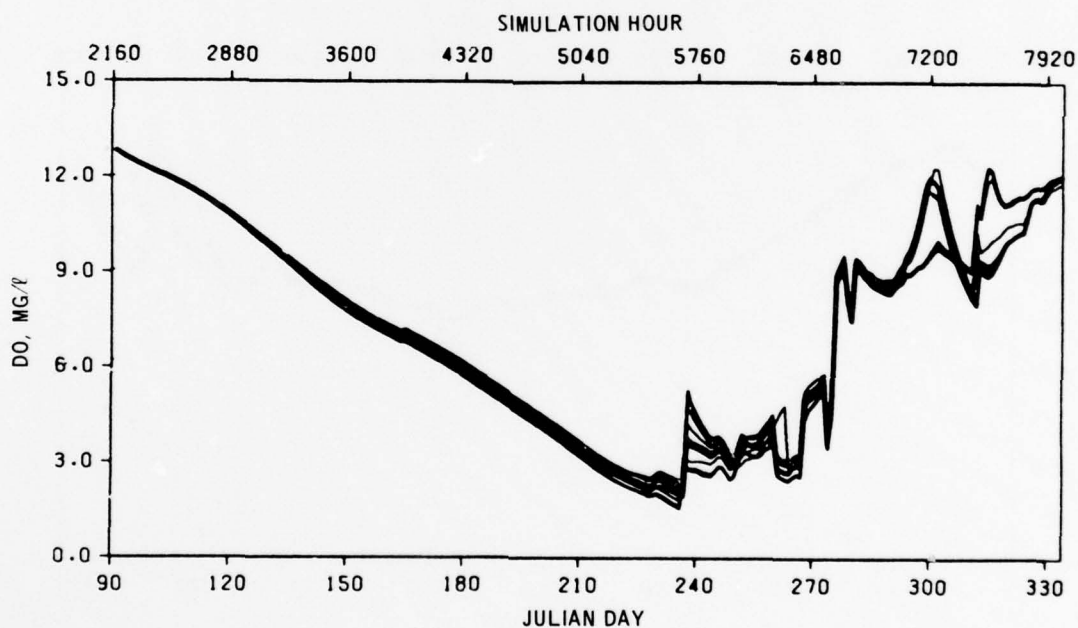


Figure 101. Monte Carlo uniform update variation simulation of DO in layer 2 (hypolimnion)



a. ITERATIONS 1-10



b. ITERATIONS 11-20

Figure 102. Monte Carlo uniform update variation simulation of DO in layer 22 (metalimnion)

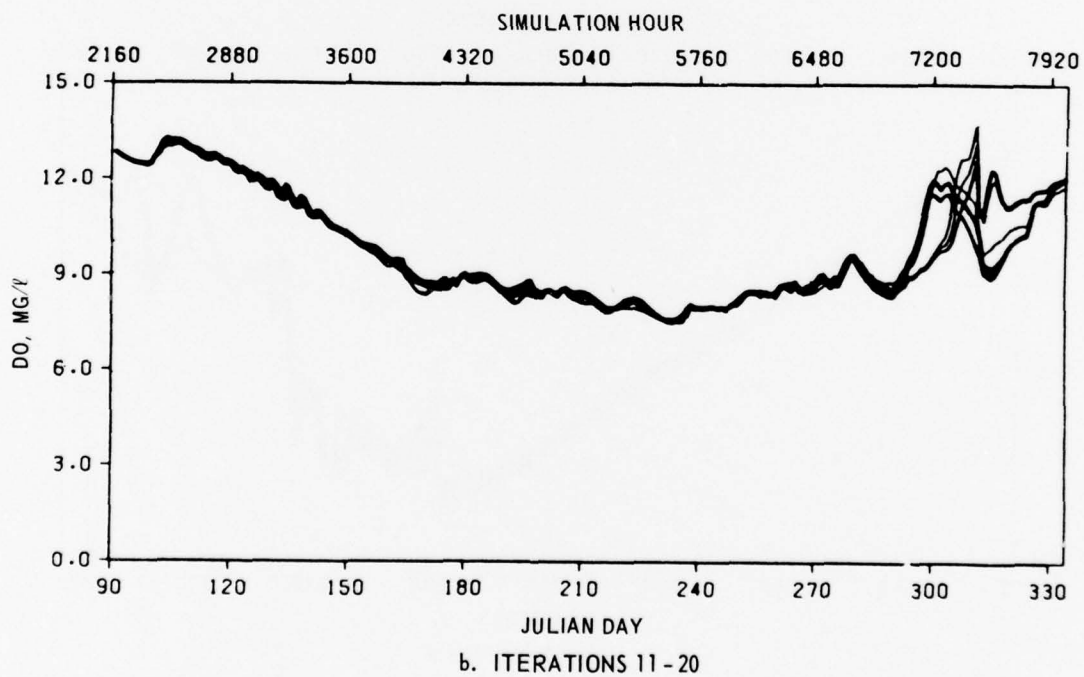
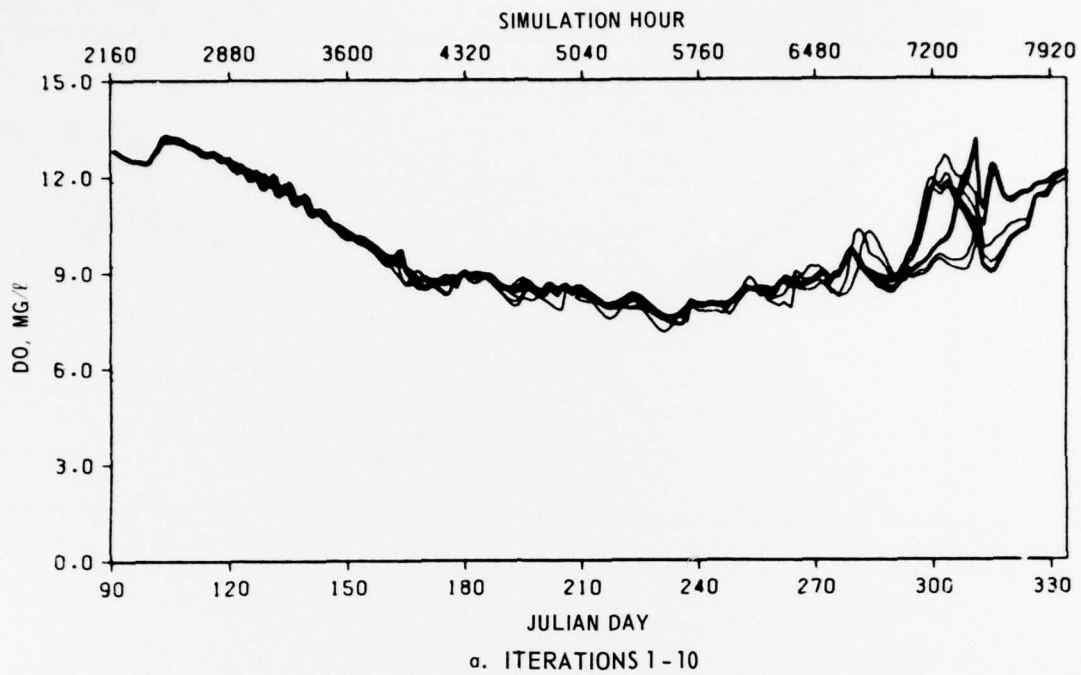


Figure 103. Monte Carlo uniform update variation simulation of DO in layer 27 (epilimnion)

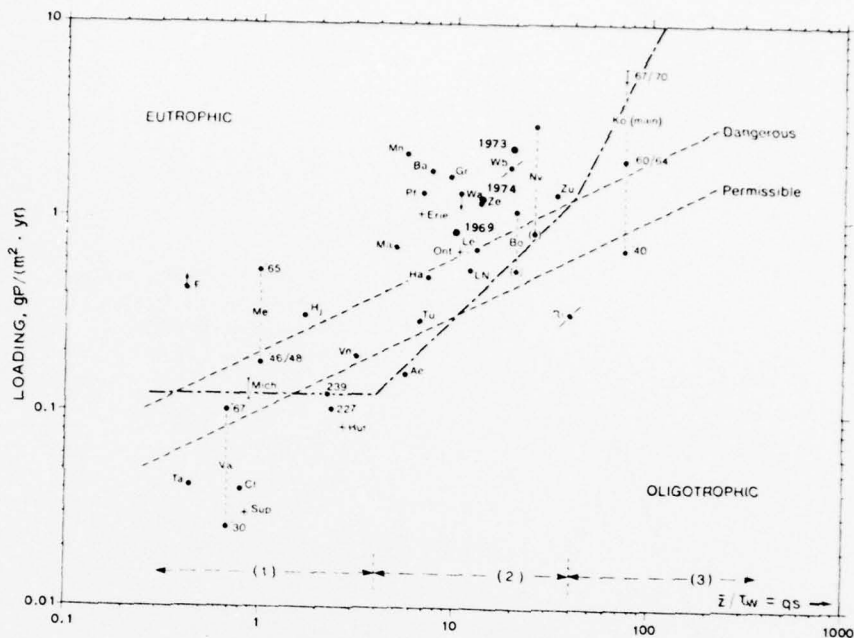


Figure 104. Determination of trophic state from phosphorus loading per unit of surface area: \bar{Z}/τ_w relationship where τ_w is the average hydraulic residence time (after Vollenweider 1975)

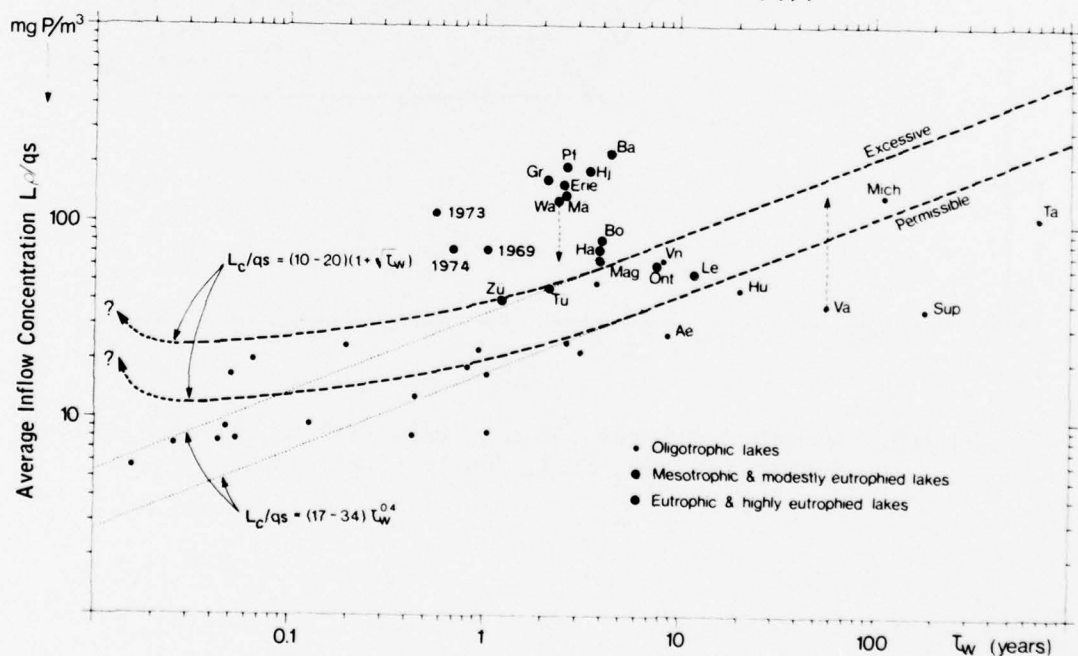


Figure 105. Determination of trophic state from average phosphorus inflow concentration: τ_w relationship (after Vollenweider 1976)

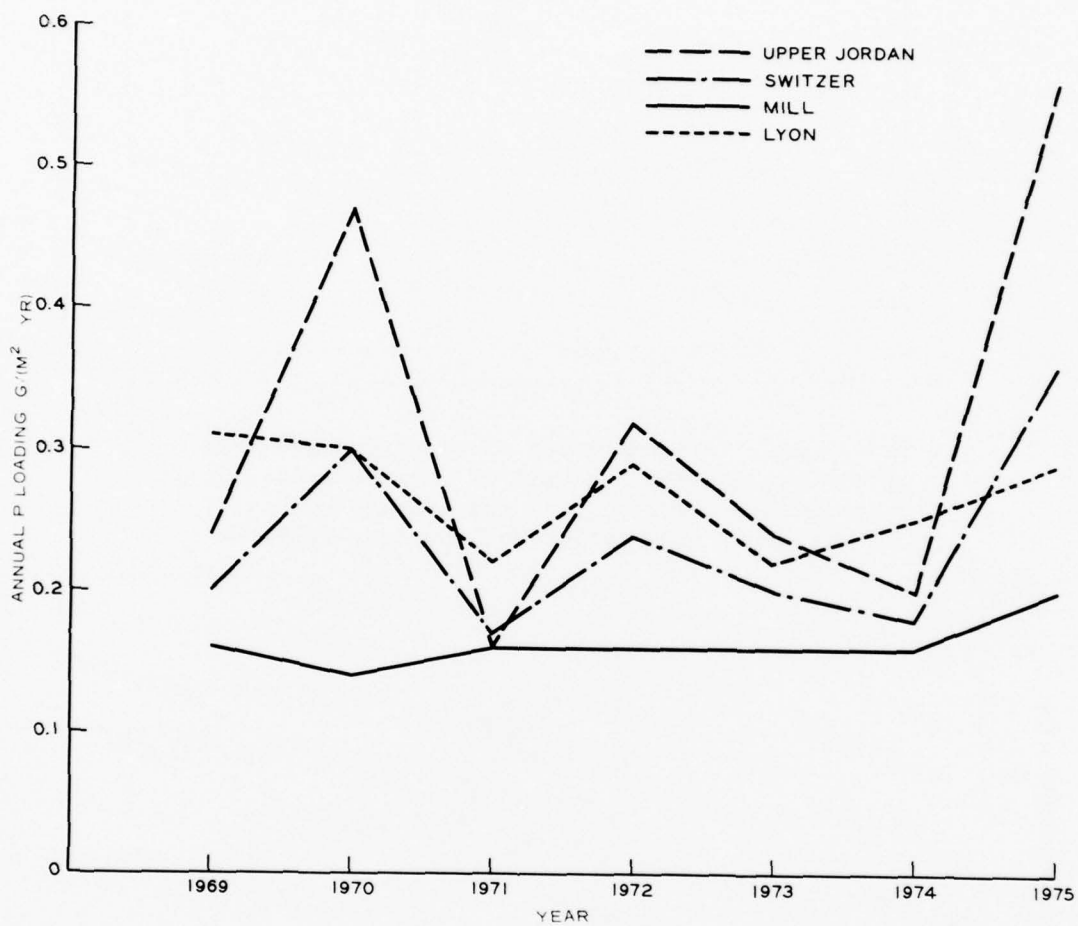


Figure 106. Annual phosphorus loadings from 1969 to 1975 for Upper Jordan, Switzer, Mill, and Lyon tributaries

APPENDIX A: REPORT ON ALGAL ASSAY PROCEDURES
BOTTLE TEST BIOASSAYS OF LAKE SAMPLES*

* Appendix A was prepared by Leslie A. Gardner, Utah Water Research Laboratory (UWRL), Utah State University, Logan, Utah 84322. UWRL is an EPA-certified laboratory through the Utah State Division of Health, Salt Lake City.

BIOASSAY RUN 1 (April 1977)

Nine samples arrived at the Utah Water Research Laboratory at 3:00 p.m. MST on March 31, 1977. All were filtered through sterile 0.45- μ Millipore filters with the exception of Belt 3, which was broken in transport.

The samples were designated by the following numbers* and will be referred to as such throughout the bioassay:

1. GS800B
2. GS800C
3. GS770
4. GS738
5. GS700
6. GS695
7. Belt 1
8. Belt 2

Upon completion of filtering, chemical analyses were performed to determine initial levels of nutrients in the samples (Table A1).

Algal bioassays were performed according to EPA (1971) using the green alga Selenastrum capricornutum PRINTZ:

Treatments (N and P additions approximate overall available nutrient concentrations initially in the sample):

- A. Sample
- B. Sample + 1.4 mg $\text{NH}_3\text{-N}/\ell$
- C. Sample + 0.124 mg $\text{PO}_4\text{-P}/\ell$
- D. Sample + NAAM levels of trace elements
- E. Sample + NAAM levels of HCO_3^-
- F. Sample + 1.4 mg $\text{NH}_3\text{-N}/\ell$ + 0.124 mg $\text{PO}_4\text{-P}/\ell$ + NAAM levels of trace elements, HCO_3^- , CaCl_2 , and MgSO_4

One and a half weeks prior to the bioassays, some S. capricornutum were placed separately into nitrogen-deficient and phosphorus-deficient NAAM. These starved algae were then placed into each of the samples making two final treatments:

* The sample locations are described in paragraph 21, main text.

G. Sample + N-starved S. capricornutum

H. Sample + P-starved S. capricornutum

Constituents of NAAM are listed in Table A2. All results are compared to a control of NAAM.

During the 18-day period of the assay, each bottle was monitored by determining the optical density (OD, Bausch and Lomb Spec 70, 750 nm, 1-cm path length). For the initial 11 days, relative fluorescence (RF x 30, Turner Fluorometer, Model 110) was utilized to monitor the progress of the cultures. Because of the difficulty of measuring biomass in low-density cultures, relative fluorescence of in vivo chlorophyll a can be used to estimate biomass during the early phases of growth. Even though relative fluorescence measures a physiological response and OD measures biomass response, the responses should correlate relative to the bioassay. This correlation can be seen in Figures A1-A16, where results for both determinations are plotted. Maximum values of OD over the test period are listed in Table A3.

OD values are linearly related to biomass as dry weight (Porcella et al. 1973). The relationship utilized to calculate volatile suspended solids (VSS) from OD (Table A4) for S. capricornutum was:

$$\text{VSS, mg/l} = 350(\text{OD}) + 3.5$$

The initial chemical analyses gave a presumptive indication that all of the samples, with the exception of GS770, were highly phosphorus limiting (Table A3). The 22.2 N/P ratio of GS770 (N/P > ~ 15 is phosphorus limiting and N/P < ~ 15 is nitrogen limiting) made it impossible to predict the limiting nutrient by chemical analysis alone.

Each of the eight samples was subjected to a routine bioassay in which separate treatments of each limiting nutrient were made (Table A1). The results of this bioassay verified the results of the original chemical analyses as indicated by the following graphs and charts.

1. GS800B

The relatively high concentrations of inorganic nitrogen in the form of NO₂ and NO₃ indicated phosphorus limitation, and bioassay

(Figures A1 and A2) verified this assumption. Because of the high concentration of $\text{NO}_2\text{-NO}_3$ in all of the samples, no NO_3 spike was used. Instead, $\text{NH}_3\text{-N}$ was substituted as the inorganic nitrogen spike due to the low level of $\text{NH}_3\text{-N}$ in the samples. The bioassay shows that the algae were unable to utilize any form of inorganic nitrogen due to the extremely low levels of phosphorus. On the other hand, the $\text{PO}_4\text{-P}$ spike (Treatment C) and the total NAAM spike (Treatment F) yielded increases of 122.85 mg/l VSS and 126.7 mg/l VSS, respectively. Such direct responses indicated definite P limitation.

Starved algae for both N and P showed no significant growth response upon addition to the sample.

2. GS800C

This sample showed a response similar to that of GS800C, although Treatment C ($\text{PO}_4\text{-P}$) and Treatment F (total NAAM) showed slightly higher increases in biomass probably due to the higher levels of N and P found in the original water sample. As with GS800B, when $\text{NH}_3\text{-N}$ was spiked, P became more limiting, and therefore, no growth was observed (Figures A3 and A4).

3. GS770

Due to the low N/P ratio (22.2) as compared to the other samples, GS770 showed a significant biomass increase in the untreated sample. Spikes of $\text{NH}_3\text{-N}$, trace metals, and HCO_3^- yielded no increase in biomass over the untreated sample, indicating a phosphorus limitation. This fact was confirmed by the increase in biomass of 84.35 mg/l VSS upon addition of $\text{PO}_4\text{-P}$ alone. It appears that GS770 will support a limited amount of growth without added nutrients but beyond a certain point becomes phosphorus limiting. Starved algae also showed a biomass increase comparable to that of the untreated sample (Figures A5 and A6).

4. GS738

The results here are somewhat disturbing due to the increase in biomass of the untreated sample while this same increase was not observed in the $\text{NH}_3\text{-N}$, trace metals, HCO_3^- , and starved algae spikes. The increase in biomass in these treatments should be comparable to the untreated sample due to the equal amount of original available nutrients.

This has been attributed to contamination at some stage in the bioassay. The $\text{PO}_4\text{-P}$ spike showed a substantial increase in growth, which confirmed P limitation for GS738 (Figures A7 and A8).

5. GS700, 6. GS695

Once again the high original total soluble inorganic nitrogen (TSIN) concentrations proved to make GS700 and GS695 phosphorus limiting as shown by bioassay as well as by chemical analysis (Figures A9 through A12).

7. Belt 1, 8. Belt 2

A slightly different situation arose concerning these two samples due to the lower initial levels of TSIN in the samples. Chemical analysis showed these samples to be P limiting (N/P ratios: Belt 1 = 146, Belt 2 = 160), but when spiked with $\text{PO}_4\text{-P}$ levels of 0.124 mg/l, both samples became N limiting (N/P ratios: Belt 1 = 3, Belt 2 = 4). Therefore, the $\text{PO}_4\text{-P}$ spike showed only a limited increase in biomass, far less than was expected. The $\text{NH}_3\text{-N}$, trace elements, HCO_3^- , and starved algae spikes showed no increase at all. The total NAAM spike gave optimum N/P ratios for maximum yield (Belt 1 = 14, Belt 2 = 15), and as would be predicted, an increase in biomass occurred. As will be noted from Table A4, biomass yield (mg/l VSS) in these two samples is much lower than observed in the other samples. This is due to the lower initial level of nutrients in Belt 1 and Belt 2. Based on these facts, it has been concluded that these two samples are also P limiting (Figures A13-A16).

Starved algae

Two sets of algae were grown one and a half weeks before the assay in NAAM, one deficient in P and the other deficient in N. These algae were then inoculated into the samples, with no other additions, and were monitored along with the other treatments.

There was no growth response in the samples with the exception of GS770(3), which had sufficient levels of both N and P to support algal growth. The maximum growth attained in each case was comparable to the growth reached in Treatment A.

Conclusions

The following conclusions were obtained from the first bioassay run:

1. All samples are P limiting, which was predicted by N/P ratios.
2. GS770 shows an N/P ratio close enough to the optimum level to give an increased biomass yield in an untreated sample, but becomes P limiting beyond a certain point as indicated by increased growth upon addition of $\text{PO}_4\text{-P}$ alone.
3. Ranking of bioassay response in untreated samples showed the following (greatest to least):

$$3 > 4 > 5 > 1 > 7 > 2 = 8 > 6$$

When the limiting nutrient was increased by spiking the ranking was:

$$3 > 2 > 1 > 6 > 4 > 5 > 8 > 7$$

4. Relative fluorescence response correlates well with OD response. This allows growth rate data to be analyzed along with biomass. Using both parameters, the N/P ratios were essentially confirmed.

5. No toxicity was observed.

6. There were no observable differences between bioassays performed with typical S. capricornutum and algae starved of a specific nutrient. This indicates that little if any nutrient carryover (excess nutrients contained in cells used for the initial inoculum) occurred.

Nutrient availability

Figures A17-A32 show maximum growth (VSS mg/l) plotted against N and P concentrations present in the various treatments. Using these figures N and P availability to the algae can be ascertained.

1. GS800B, 2. GS800, 3. GS770,
4. GS738, 5. GS700, 6. GS695

In each of these samples, yields (mg VSS/mg $\text{PO}_4\text{-P}$) were equal or higher with the addition of P alone as compared to a control of algae grown in NAAM. These results indicate that all the $\text{PO}_4\text{-P}$ in each

treatment was used to produce biomass. The addition of N alone created no increase in biomass. In each case, the addition of N resulted in yields equal to or less than yields in untreated samples. P was the limiting nutrient in these samples.

7. Belt 1, 8. Belt 2

Addition of neither P nor N alone cause any increased yield in biomass. When both nutrients were added to the samples, biomass increased but not to the level of algae grown in NAAM. This may indicate a slight amount of toxicity. Both N and P are limiting nutrients in these samples.

BIOASSAY RUN 2 (June 1977)

Six samples arrived at the Utah Water Research Laboratory on Thursday, June 23, at 3:00 p.m. MDT. Belt 1 and Belt 2 were eliminated from this bioassay. Samples were filtered and analyzed in a manner identical to that in Run 1. The samples were designated the same numbers, and treatments were exactly as before (pages A2 and A3).

The results of the chemical analyses are listed in Table A5. As was true of Run 1, the chemical analyses showed each of the samples to be phosphorus limiting, with the exception of GS770 (N/P ratios, Table A6). Bioassay verified this assumption, but a slightly different set of circumstances arose in Run 2, which were not seen in Run 1.

1. GS800B, 2. GS800C,
4. GS738, 5. GS700, 6. GS695

These samples will be discussed together because all of them show responses similar in nature. The bioassays definitely indicated phosphorus limitation in all cases; but as is shown by the OD figures, the response to addition of $\text{PO}_4\text{-P}$ only (Treatment C) is somewhat less than that to total NAAM (Treatment F) (Table A7). The assumption to be made here is that at the point the growth in Treatment C leveled off was where some other nutrient became limiting. It seems highly possible that nitrogen became the limiting factor. Bear in mind that any of the NAAM constituents (HCO_3^- , trace metals, CaCl_2 , MgSO_4 , or N) may have become limiting. It would require that each of these constituents be added to a $\text{PO}_4\text{-P}$ spike separately in order to definitely ascertain the

limiting factor. It is only assumed that N became limiting because upon addition of $\text{PO}_4\text{-P}$ to the samples, the N/P ratio was lowered to a point that may have been less than optimum for maximum growth (Figures A33-A42).

3. GS770

As before, GS770 contained initial nutrient levels that supported increases in biomass without further addition of nutrients. In fact, the nutrient levels (Table A6) were so much greater that addition of total NAAM caused no further increase in biomass over the time period the bioassay was monitored. It will be noticed that all other spikes actually brought about a decrease in the amount of growth as compared to the untreated sample. It is possible that some precipitation of nutrients occurred in these treatments (Figures A43 and A44).

Starved algae

No growth response was observed when the samples were inoculated with nutrient-starved algae. Once again this did not hold true for 3 (GS770), in which growth for both P and N starved algae approached that found in samples inoculated with typical S. capricornutum.

Conclusions

The following conclusions were obtained from the second bioassay run:

1. All samples, with the exception of 3 (GS770), are P limiting as indicated by bioassay and chemical analysis.
2. The high levels of nitrogen and phosphorus found in GS770 at the time of analysis indicate that this sample is capable of supporting algal growth without further additions of nutrients. There does not appear to be a limiting factor at the levels and over the time period measured.
3. Ranking of the bioassay response in untreated samples showed the following (greatest to least):

$$3 > 6 > 2 = 4 > 1 > 5$$

When the limiting nutrient was increased by spiking, the ranking was:

$$1 > 4 > 5 > 2 > 6$$

3 appears to have no limitation.

4. No toxicity was observed.

5. Algae starved of a specific nutrient showed no differences in growth response over typical S. capricornutum.

Nutrient availability

1. GS800B, 2. GS800C,

4. GS738, 5. GS700, 6. GS695

Each of these samples showed similar response to addition of nutrients (Figures A45-A56). Addition of P caused an increased yield in biomass, but in all samples the yield was less than that seen in NAAM. Addition of N caused no appreciable increase in biomass over the untreated samples. When both N and P were placed in the samples, growth equaled that of NAAM. This clearly indicates that P is the limiting nutrient, but N becomes limiting before maximum growth is reached. It appears that nutrients are fully utilized for growth.

3. GS770

The initial levels of nutrients in this sample were great enough to enable optimum growth in untreated samples. When more N or P or both were added, yields actually decreased. This indicates that N and P may have become toxic at these high levels, and therefore growth decreased.

REFERENCES

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- Porcella, D. B., P. A. Cowan, and E. J. Middlebrooks. 1973. Biological Response to Detergent and Non-detergent Phosphorus in Sewage - Part II. Water and Sewage Works, December. 5 pages.
- Solorzano, L. 1969. Determination of Ammonia in Natural Waters by the Phenolhypochlorite Method. Limnology and Oceanography 14(5):799-801.
- Standard Methods for the Examination of Water and Wastewater. 1976. U. S. Public Health Service. 14th Edition. 874 pages.
- Strickland, J. D. H., and T. R. Parsons. 1968. A Practical Handbook of Seawater Analyses. Fisheries Research Board of Canada, Ottawa. 311 pages.

Table A1. Chemical analyses and results, Run 1.^a

Parameter	Units	Method	Refs.*	GS800B	GS800C	GS770	GS738	GS700	GS695	Belt 1	Belt 2
PO ₄ -P	µg P/ℓ	Antimony Molybdate; Ascorbic Acid	1	5	6	55	5	3	4	3	3
Total Soluble P (TSP)	µg P/ℓ	Persulfate Digestion	1	18	8	75	7	8	11	9	8
(NO ₃ +NO ₂)-N	µg N/ℓ	Cadmium Reduction; Diazotization	2	1008	1117	1138	968	1328	891	417	457
NH ₃ -N	µg N/ℓ	Indophenol	3	22	24	83	32	33	18	21	24
Total Soluble Inorganic N (TSIN)	µg N/ℓ			1030	1141	1221	1000	1361	909	438	481
Total Kjeldahl N	µg N/ℓ	Digestion Distillation	4	22	24	83	32	33	47	22	141
Total Nitrogen	µg N/ℓ			1030	1141	1221	1000	1361	938	439	598

- * 1. Standard Methods, 1976.
 2. Strickland and Parsons, 1968.
 3. Solorzano, 1969.
 4. Environmental Protection Agency, 1974.

^aUWRL is an EPA-certified laboratory through the Utah State Division of Health, Salt Lake City.

Table A2. Nutrient algal assay medium (NAAM).

Compound		Concentration in NAAM	
		Compound mg/l	Element mg/l
A ₁	NaNO ₃	25.500	N 4.2
A ₂	MgCl ₂ ·6H ₂ O	12.171	Mg 2.9
	MgSO ₄ ·7H ₂ O	14.700	
A ₃	CaCl ₂ ·2H ₂ O	4.410	Ca 1.2
A ₄	NaHCO ₃	15.000	
B	K ₂ HPO ₄	1.044	P 0.186
		<u>µg/l</u>	<u>µg/l</u>
C	H ₃ BO ₃	185.64	B 32.45
	MnCl ₂ ·4H ₂ O	417.18	Mn 115.80
	ZnCl ₂	32.70	Zn 15.68
	Na ₂ MoO ₄ ·2H ₂ O	7.26	Mo 2.88
	CoCl ₂ ·6H ₂ O	1.43	Co 0.35
	CuCl ₂ ·2H ₂ O	0.01	Cu 0.004
D	FeCl ₃ ·6H ₂ O	160	Fe 33.05
	Na ₂ EDTA·2H ₂ O	300	
			<u>mg/l</u>
All			S 1.91
			Na 11.04
			K 0.47
			C 2.14

Protocol for Nutrient Spiking

Compound	Nutrient
A ₁	Nitrogen
B	Phosphorus
A ₁ + B	N + P
C + D	Trace Elements (T.E.)
All	NAAM

Reference: U. S. Environmental Protection Agency (1971)

Table A3. Maximum amount of growth (day), 750 nm, 1 cm OD, Run 1. Value in parentheses is the earliest day on which the greatest OD value was observed.

Sample	Nutrient Starved S. cap.								N/P ^a Ratios TSIN/ OP TSP	
	Sample Treat- ment A (Sample Only)	Sample Treat- ment B (NH ₃ -N)	Sample Treat- ment C (PO ₄ -P)	Sample Treat- ment D (Trace Elements)	Sample Treat- ment E (HCO ₃ ⁻)	Sample Treat- ment F (b)	Sample			
							Treat- ment G (N- Starved)	Treat- ment H (P- Starved)		
1.GS800B	0.026(18)	0.017(3)	0.377(18)	0.026(3)	0.016(3)	0.398(15)	0.017(3)	0.011(9)	206	57
2.GS800C	0.015(9)	0.021(3)	0.422(18)	0.020(3)	0.017(9)	0.384(18)	0.019(3)	0.015(3)	190	143
3.GS770	0.239(15)	0.221(11)	0.480(15)	0.174(15)	0.181(15)	0.569(18)	0.221(15)	0.235(15)	22	16
4.GS738	0.210(18)	0.024(18)	0.340(18)	0.019(3)	0.015(3)	0.353(18)	0.019(3)	0.014(3)	200	143
5.GS700	0.089(18)	0.019(3)	0.334(18)	0.014(3)	0.013(3)	0.353(15)	0.021(3)	0.008(9)	454	170
6.GS695	0.010(9)	0.010(18)	0.364(18)	0.012(9)	0.013(5)	0.338(16)	0.035(5)	0.005(3)	227	83
7.Belt 1	0.022(12)	0.008(3)	0.049(16)	0.009(18)	0.010(18)	0.181(16)	0.010(18)	0.009(18)	146	49
8.Belt 2	0.015(9)	0.011(9)	0.074(16)	0.009(9)	0.010(9)	0.257(18)	0.011(9)	0.008(5)	160	60
NAAM	0.519(18)						0.480(18)	0.475(18)		

^aN/P = Nitrogen/phosphorus ratios for original water samples.

^b1.4 mg/l NH₃-N + 0.124 mg/l PO₄ + NAAM levels of trace elements, HCO₃⁻, CaCl₂, and MgSO₄.

TSIN = Total Soluble Inorganic Nitrogen

OP = Orthophosphate

TSP = Total Soluble Phosphorus

Table A4. Maximum amount of growth (day), mg/l VSS, Run 1.^a Value in parentheses is the earliest day on which the greatest OD value was observed.

Sample	Nutrient Starved									
	S. cap.									
	Sample Treatment (Sample Only)	Sample Treatment (NH ₃ -N)	Sample Treatment (PO ₄ -P)	Sample Treatment (Trace Elements)	Sample Treatment (HCO ₃ ⁻)	Sample Treatment F (b)	Sample Treatment G (N-Starved)	Sample Treatment H (P-Starved)		
1.GS800B	12.6(18)	9.45(3)	135.45(18)	12.6(3)	9.1(3)	142.8(15)	9.45(3)	7.35(9)		
2.GS800C	9.45(9)	10.85(3)	151.20(18)	10.5(3)	9.45(9)	137.90(18)	10.15(3)	8.75(3)		
3.GS770	87.15(15)	80.85(11)	171.50(15)	64.4(15)	66.85(15)	202.65(18)	80.85(15)	85.75(15)		
4.GS738	77.0(18)	11.90(18)	122.50(18)	10.15(3)	8.75(3)	127.05(18)	10.15(3)	8.4(3)		
5.GS700	34.65(18)	10.15(3)	120.40(18)	8.40(3)	8.05(3)	127.05(15)	10.85(3)	6.3(9)		
6.GS695	7.00(9)	7.00(18)	130.90(18)	7.7(9)	8.05(5)	121.90(16)	15.75(5)	5.25(3)		
7.Belt 1	11.20(12)	6.3(3)	20.65(16)	6.65(18)	7.00(18)	66.85(16)	7.00(18)	6.65(18)		
8.Belt 2	8.75(9)	7.35(9)	29.40(16)	6.65(9)	7.00(9)	93.45(18)	7.35(9)	6.30(5)		
NAAM	185.15(18)						171.50(18)	169.75(18)		

^aVSS = Volatile Suspended Solids.

VSS, mg/l = 350 (OD) + 3.5 (Porcella et al., 1973).

^b1.4 mg/l NH₃-N + 0.124 mg/l PO₄ + NAAM levels of trace elements, HCO₃⁻, CaCl₂, and MgSO₄.

Table A5. Chemical analyses and results, Run 2.

Parameter	Units	Method	Refs.*	1 GS800B	2 GS800C	3 GS770	4 GS738	5 GS700	6 GS695
PO ₄ -P	µg P/l	Antimony Molybdate Ascorbic Acid	1	9	8	153	4	4	11
Total Soluble P (TSP)	µg P/l	Persulfate Digestion	1	12	13	176	40	8	11
(NO ₃ -NO ₂)-N	µg N/l	Cadmium Reduction; Diazotization	2	1960	1820	3084	2047	2120	2360
NH ₃ -N	µg N/l	Indophenol	3	10	12	32	8	10	32
Total Soluble Inorganic N (TSIN)	µg N/l			1970	1832	3116	2055	2120	2392
Total Kjeldahl N	µg N/l	Digestion Distillation	4	78	156	141	86	63	32
Total Nitrogen	µg N/l			2038	1976	3175	2133	2183	2392

* 1. Standard Methods, 1976.

2. Strickland and Parsons, 1968.

3. Solorzano, 1969.

4. Environmental Protection Agency, 1974

Table A6. Maximum amount of growth (day), 750 nm, 1 cm OD, Run 2. Value in parentheses is the earliest day on which the greatest OD value was observed.

Sample	Nutrient Starved S. cap.									
	Sample Treat- ment A (Sample Only)	Sample Treat- ment B (NH ₃ -N)	Sample Treat- ment C (PO ₄ -P)	Sample Treat- ment D (Trace Elements)	Sample Treat- ment E (HCO ₃ ⁻)	Sample Treat- ment F (b)	Sample Treat- ment G (N- Starved)	Sample Treat- ment H (P- Starved)	N/P ^a Ratios TSIN/ OP TSP	
1.GS800B	0.020(7)	0.025(7)	0.202(13)	0.027(13)	0.019(7)	0.311(13)	0.027(3)	0.014(3)	219	164
2.GS800C	0.022(5)	0.022(5)	0.153(13)	0.028(13)	0.022(5)	0.273(13)	0.034(3)	0.014(3)	229	141
3.GS770	0.338(13)	0.278(13)	0.224(13)	0.268(13)	0.323(13)	0.349(13)	0.263(13)	0.209(13)	20	18
4.GS738	0.022(7)	0.012(7)	0.180(13)	0.016(7)	0.015(7)	0.286(13)	0.016(5)	0.009(13)	514	51
5.GS700	0.018(5)	0.014(5)	0.173(13)	0.014(5)	0.015(5)	0.258(13)	0.027(5)	0.010(13)	530	265
6.GS695	0.025(5)	0.022(5)	0.084(5)	0.024(5)	0.029(7)	0.258(13)	0.038(7)	0.026(5)	217	217

^aN/P = Nitrogen/phosphorus ratios for original water samples.

^b1.4 mg/l NH₃-N + 0.124 mg/l PO₄ + NAAM levels of trace elements, HCO₃⁻, CaCl₂, and MgSO₄.

TSIN = Total Soluble Inorganic Nitrogen

OP = Orthophosphate

TSP = Total Soluble Phosphorus

Table A7. Maximum amount of growth (day), mg/l VSS, Run 2.^a Value in parentheses is the earliest day on which the greatest OD value was observed.

Sample	Sample Treatment A (Sample Only)	Sample Treatment B (NH ₃ -N)	Sample Treatment C (PO ₄ -P)	Sample Treatment D (Trace Elements)	Sample Treatment E (HCO ₃ ⁻)	Sample Treatment F (b)	Nutrient Starved		
							Sample Treatment G (N-Starved)	Sample Treatment H (P-Starved)	S. cap.
1.CS800B	10.5(7)	12.25(7)	74.2(13)	12.95(13)	10.15(7)	112.35(13)	12.95(3)	8.40(3)	
2.CS800C	11.2(5)	11.20(5)	57.05(13)	13.30(13)	11.20(5)	99.05(13)	28.35(3)	8.40(3)	
3.CS770	121.8(13)	100.8(13)	81.90(13)	97.30(13)	116.55(13)	125.65(13)	95.55(13)	76.65(13)	
4.CS738	11.2(7)	7.7(7)	66.50(13)	9.10(7)	8.75(7)	103.60(13)	9.10(5)	6.65(13)	
5.CS700	9.8(5)	8.4(5)	64.05(13)	8.40(5)	8.75(5)	93.80(13)	12.95(5)	7.00(13)	
6.CS695	12.25(5)	11.2(5)	32.90(5)	11.90(5)	13.65(7)	93.80(13)	16.80(7)	12.60(5)	

^aVSS = Volatile Suspended Solids.

VSS, mg/l = 350 (OD) + 3.5 (Porcella et al., 1973).

^b1.4 mg/l NH₃-N + 0.124 mg/l PO₄ + NAAM levels of trace elements, HCO₃⁻, CaCl₂, and MgSO₄.

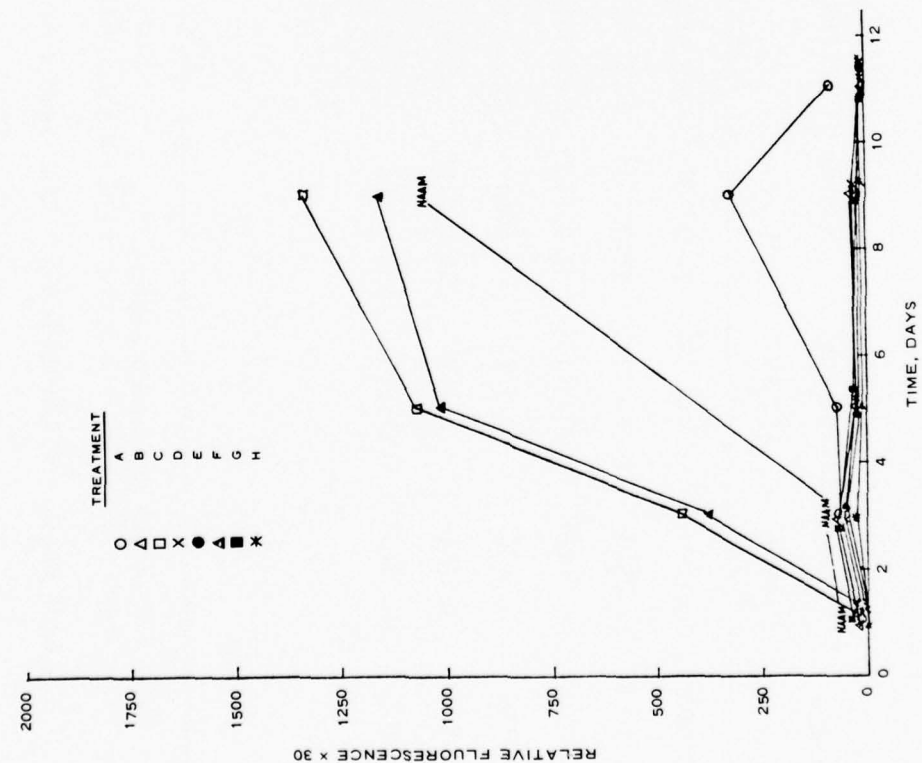


Figure A1. Relative fluorescence of sample GS800B with various treatments

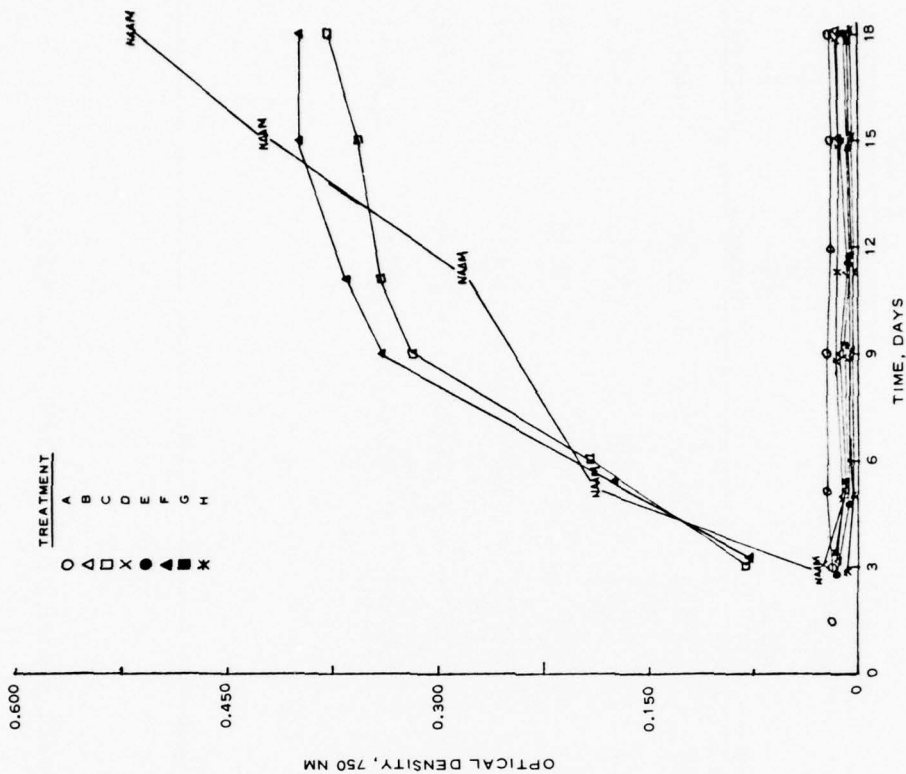


Figure A2. Optical density of sample GS800B with various treatments

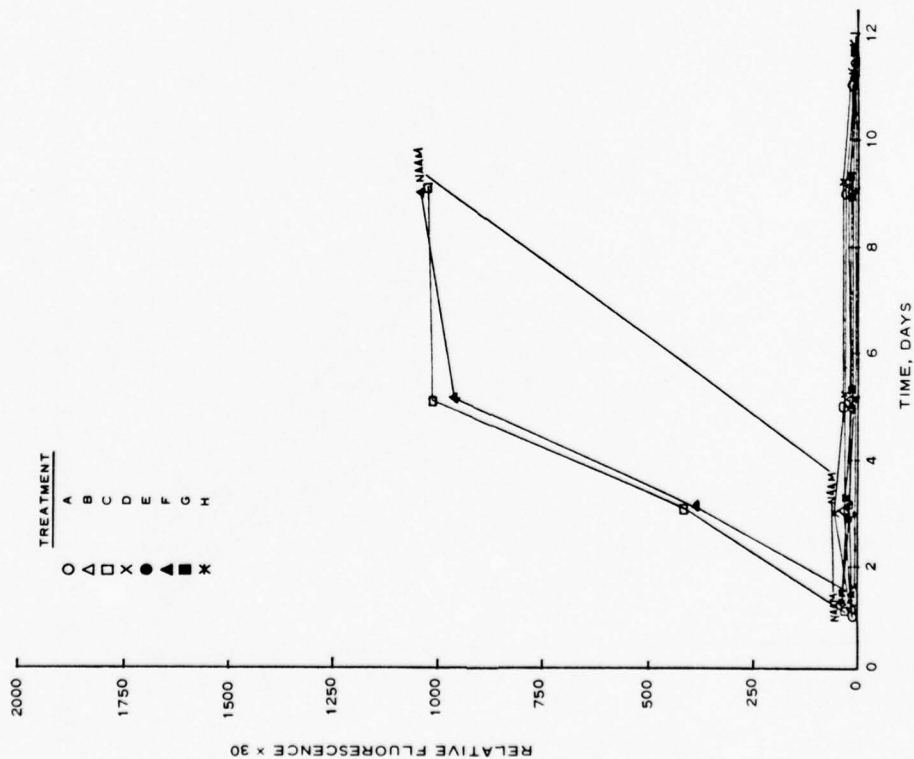


Figure A3. Relative fluorescence of sample GS800C with various treatments

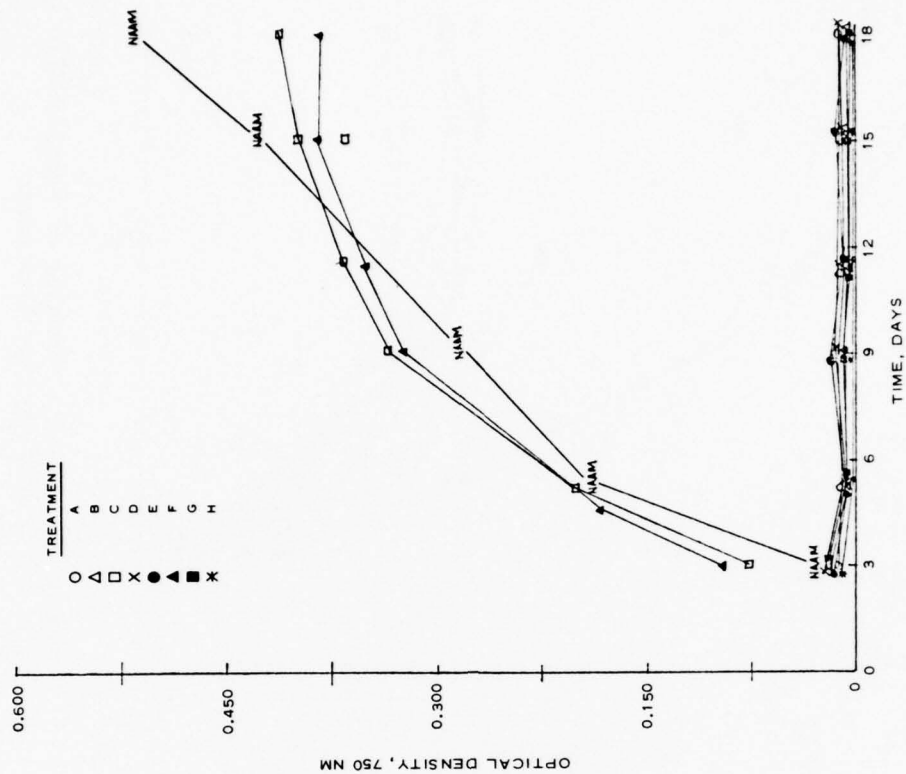


Figure A4. Optical density of sample GS800C with various treatments

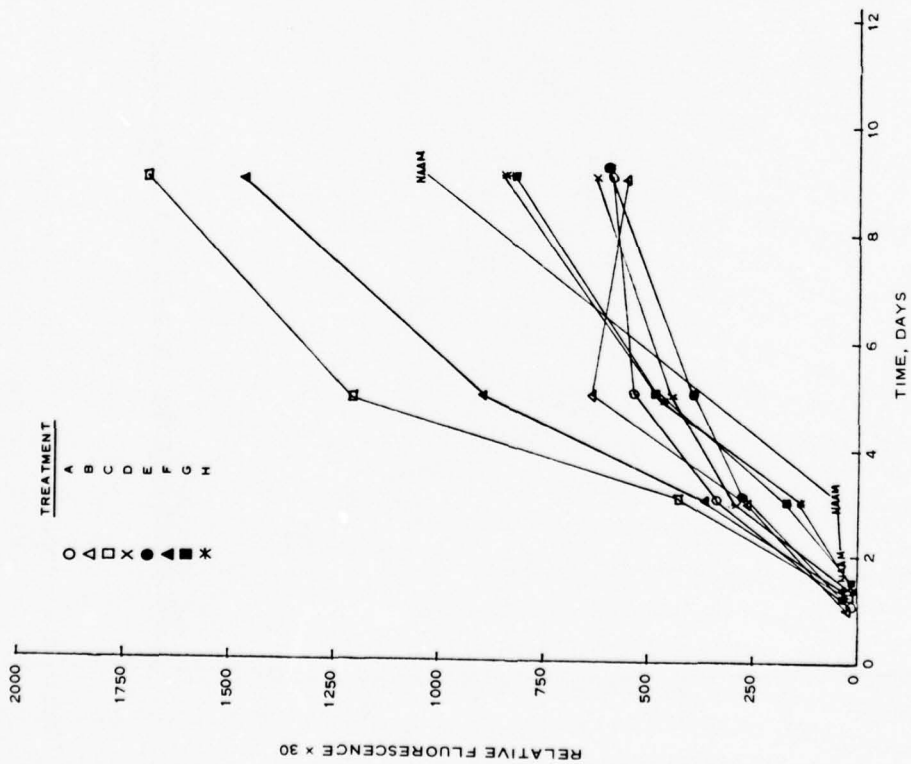


Figure A5. Relative fluorescence of sample GS770 with various treatments

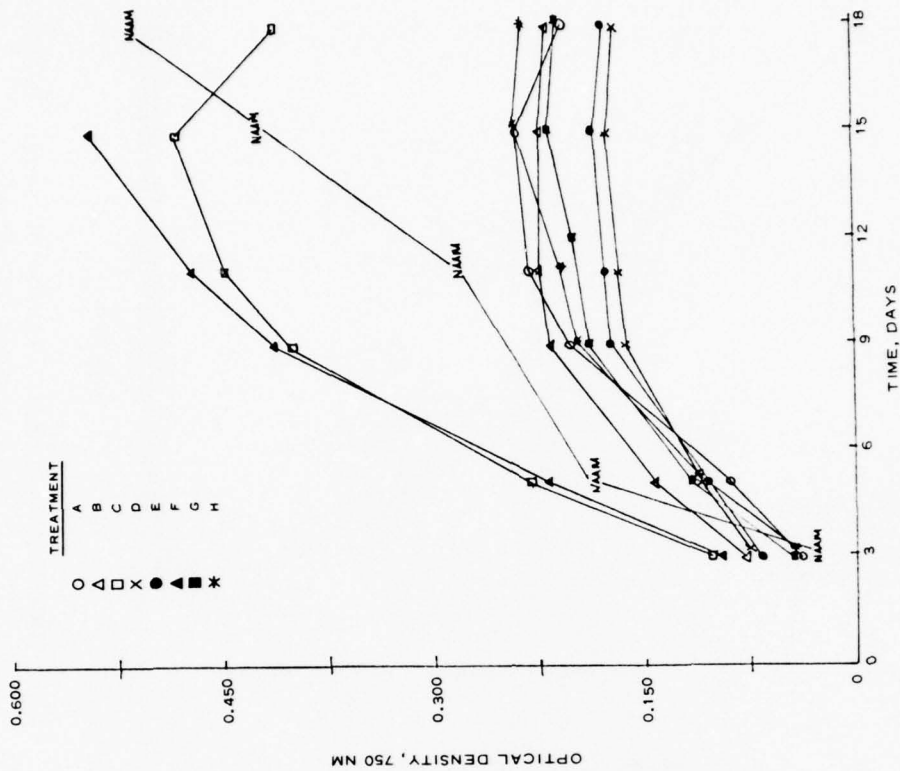


Figure A6. Optical density of sample GS770 with various treatments

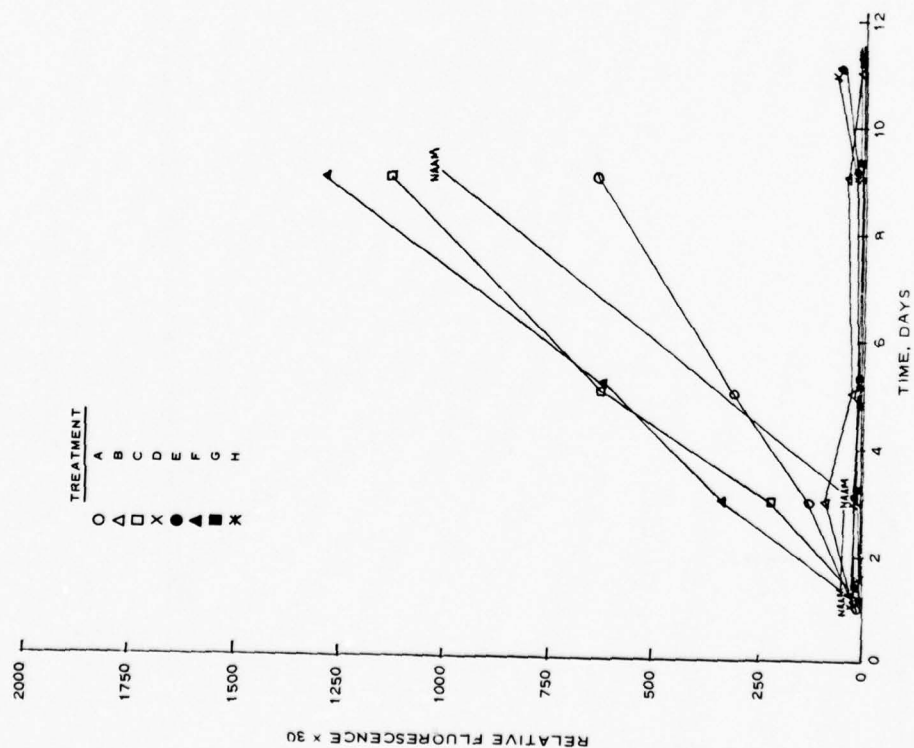


Figure A7. Relative fluorescence of sample GS738 with various treatments

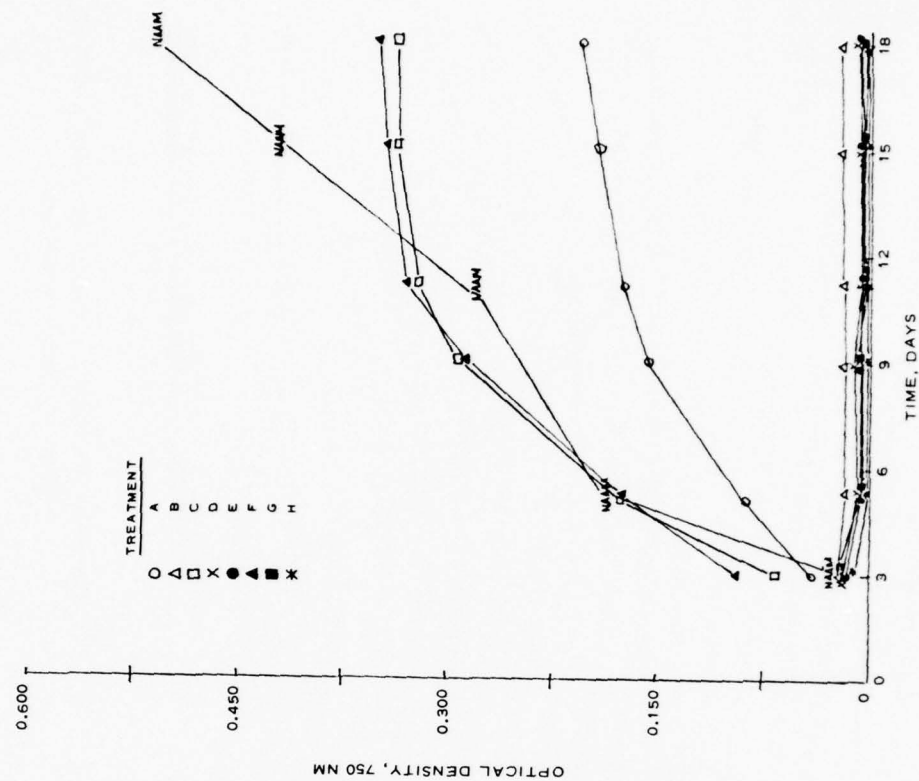


Figure A8. Optical density of sample GS738 with various treatments

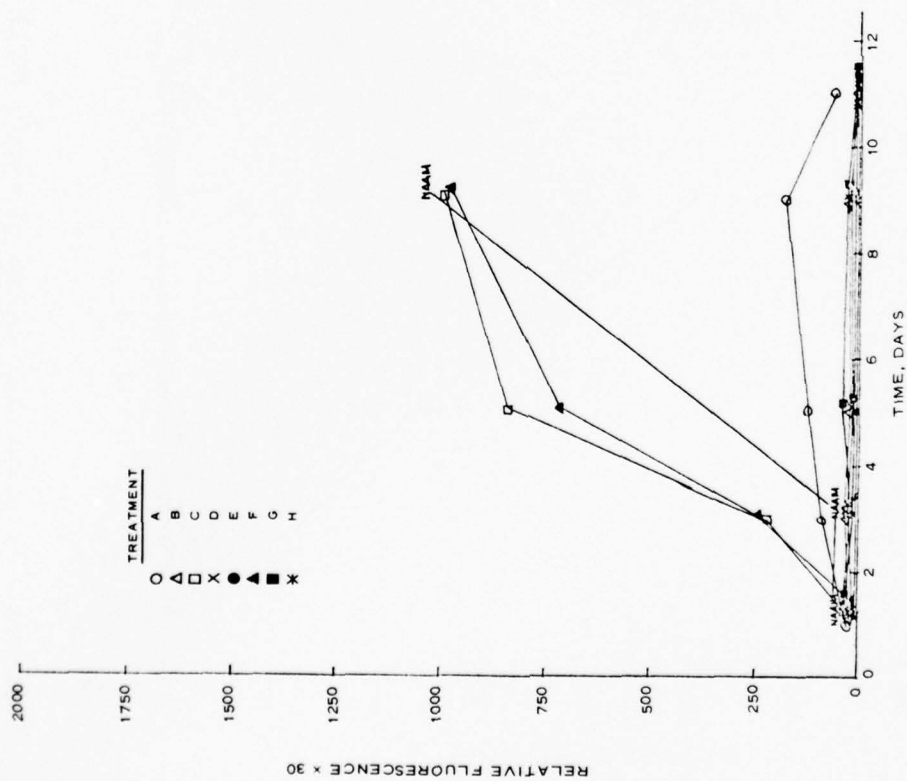


Figure A9. Relative fluorescence of sample GS700 with various treatments

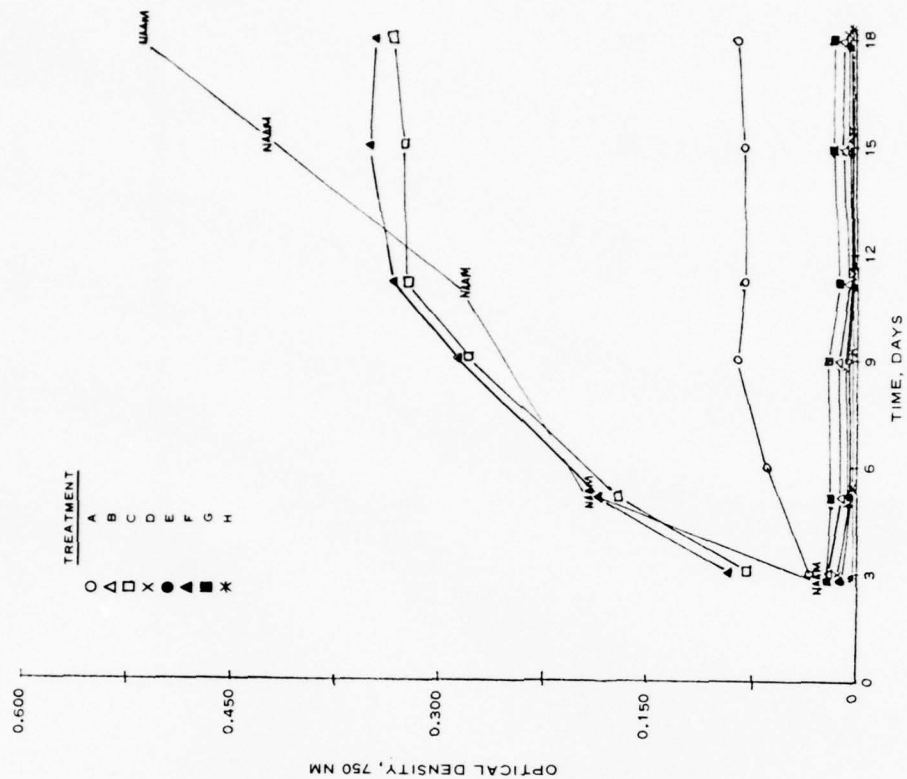


Figure A10. Optical density of sample GS700 with various treatments

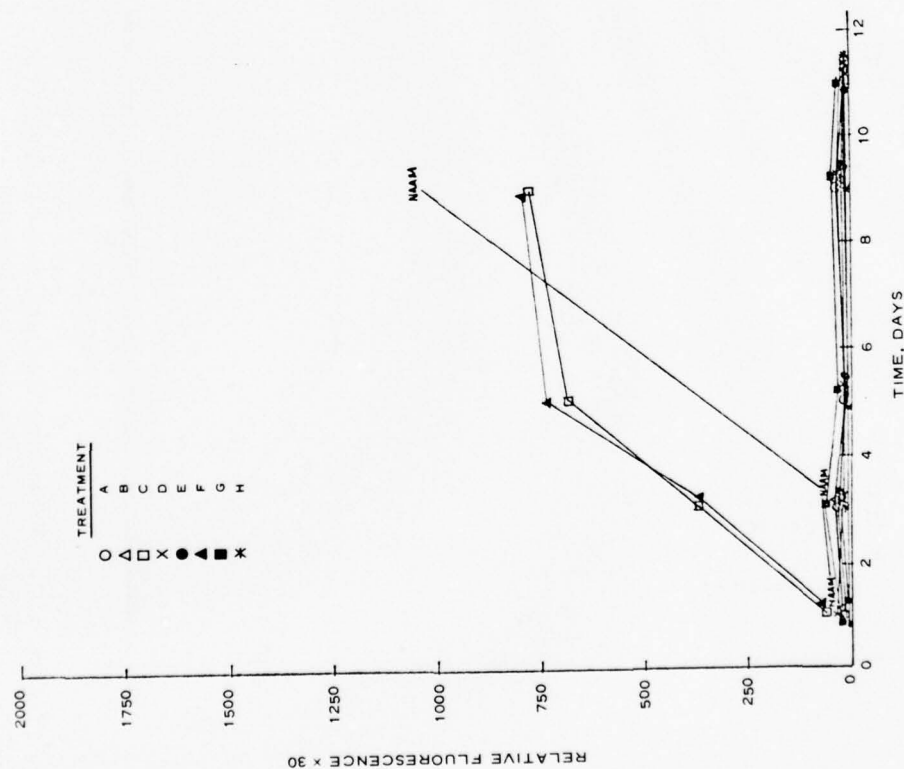


Figure A11. Relative fluorescence of sample GS695 with various treatments

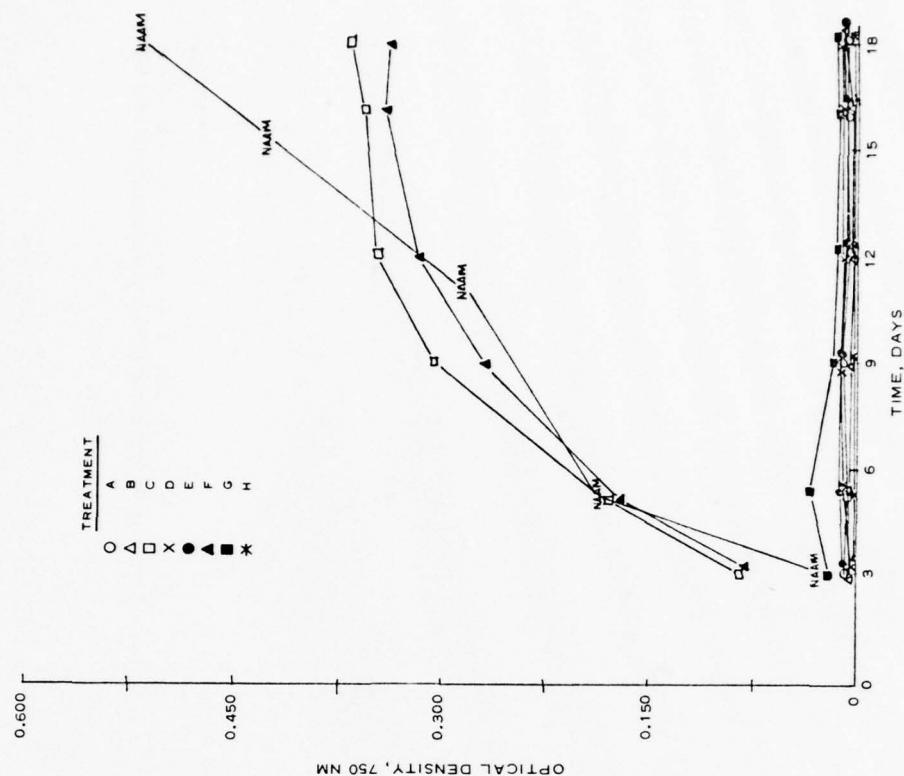


Figure A12. Optical density of sample GS695 with various treatments

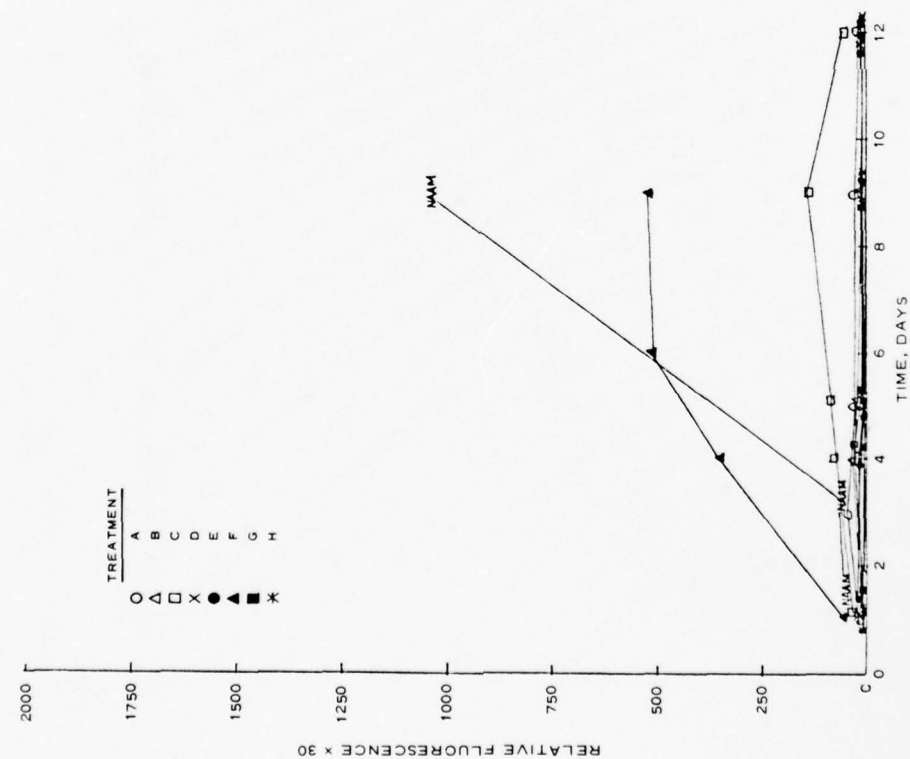


Figure A13. Relative fluorescence of sample Belt 1 with various treatments

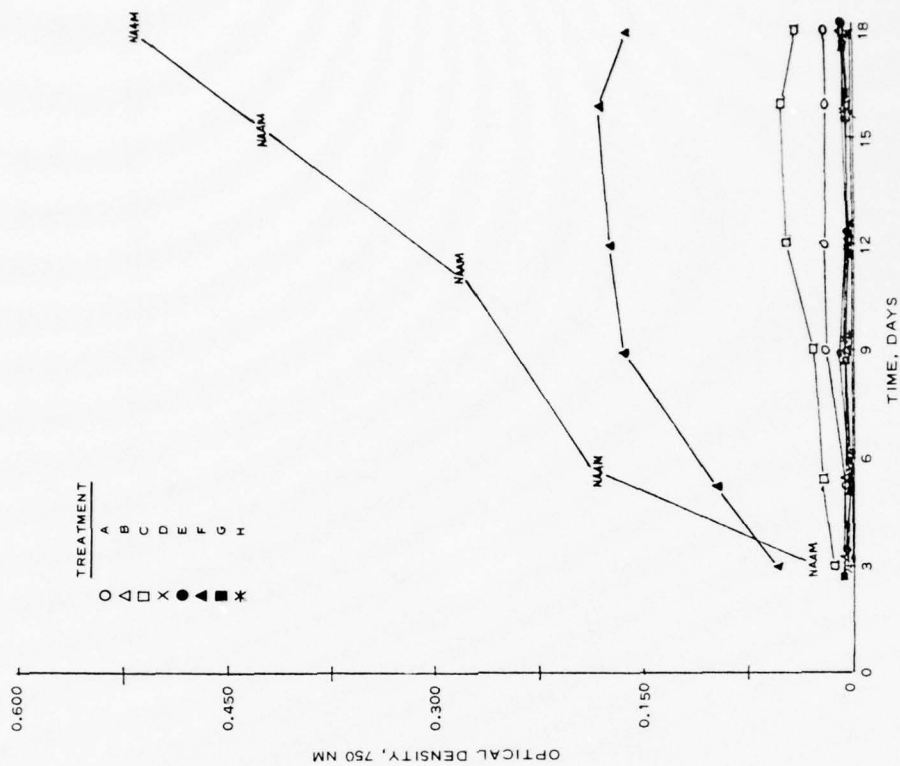


Figure A14. Optical density of sample Belt 1 with various treatments

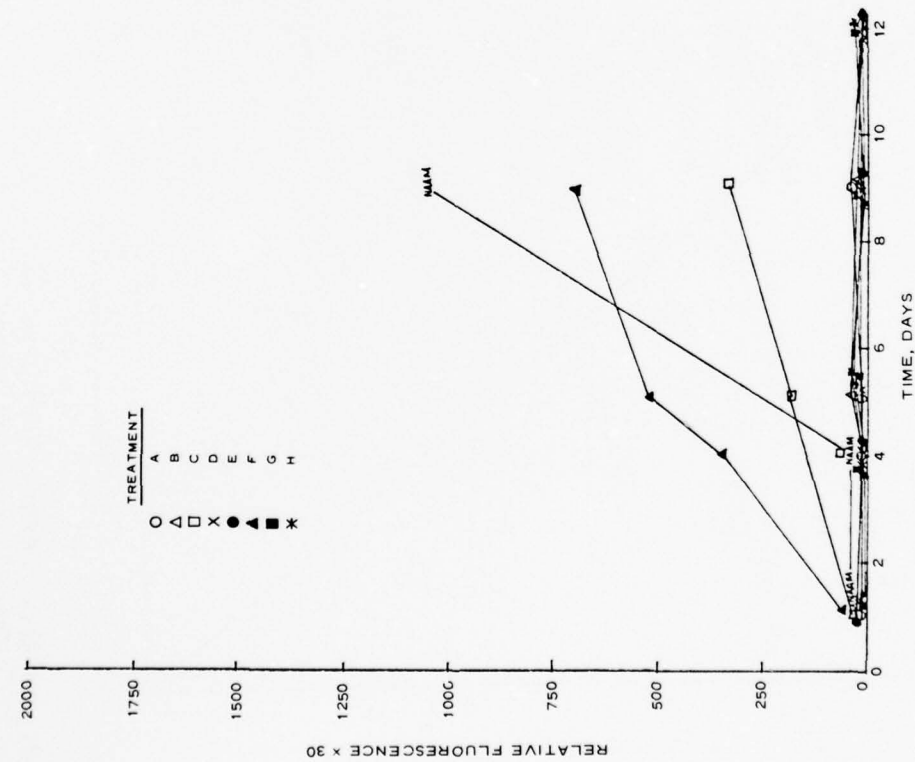


Figure A15. Relative fluorescence of sample Belt 2 with various treatments

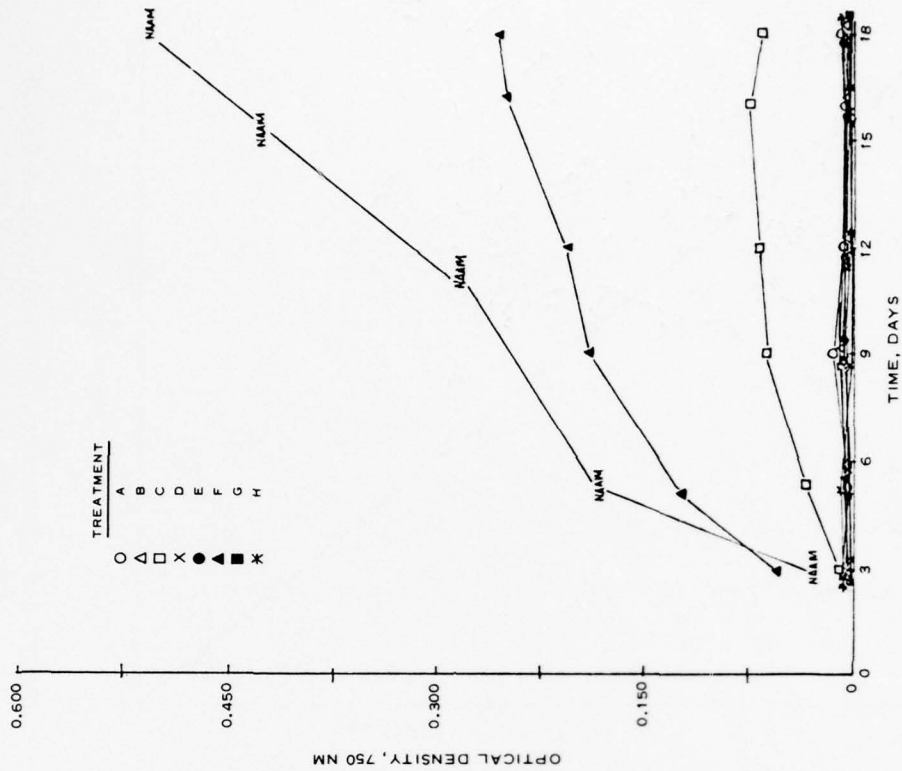


Figure A16. Optical density of sample Belt 2 with various treatments

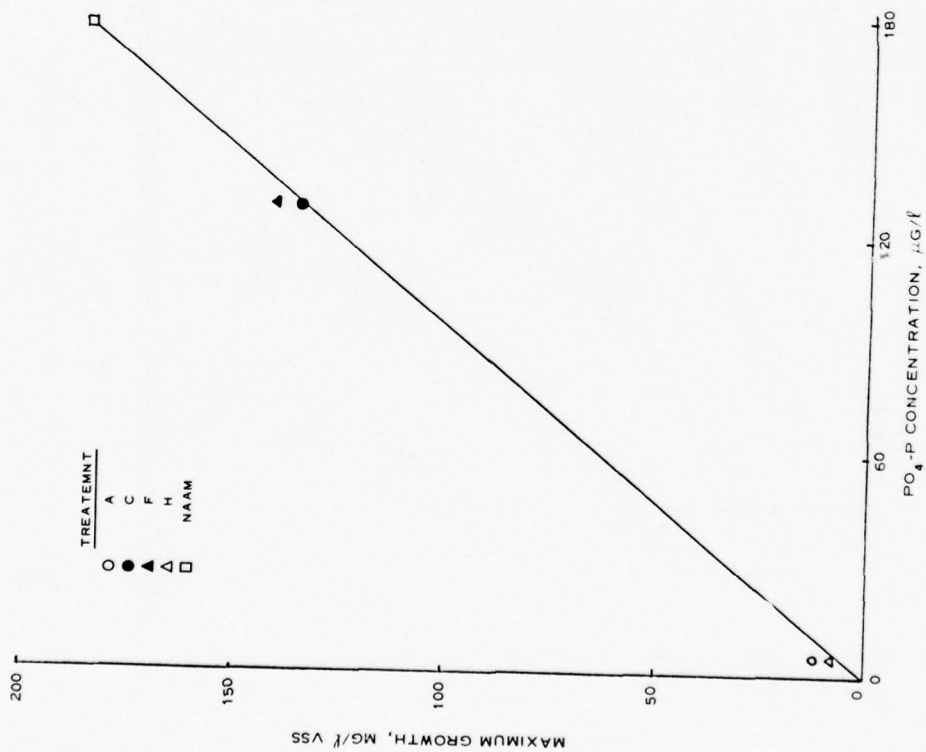


Figure A17. Maximum growth in sample GS800B (Run 1) as a function of $PO_4\text{-P}$

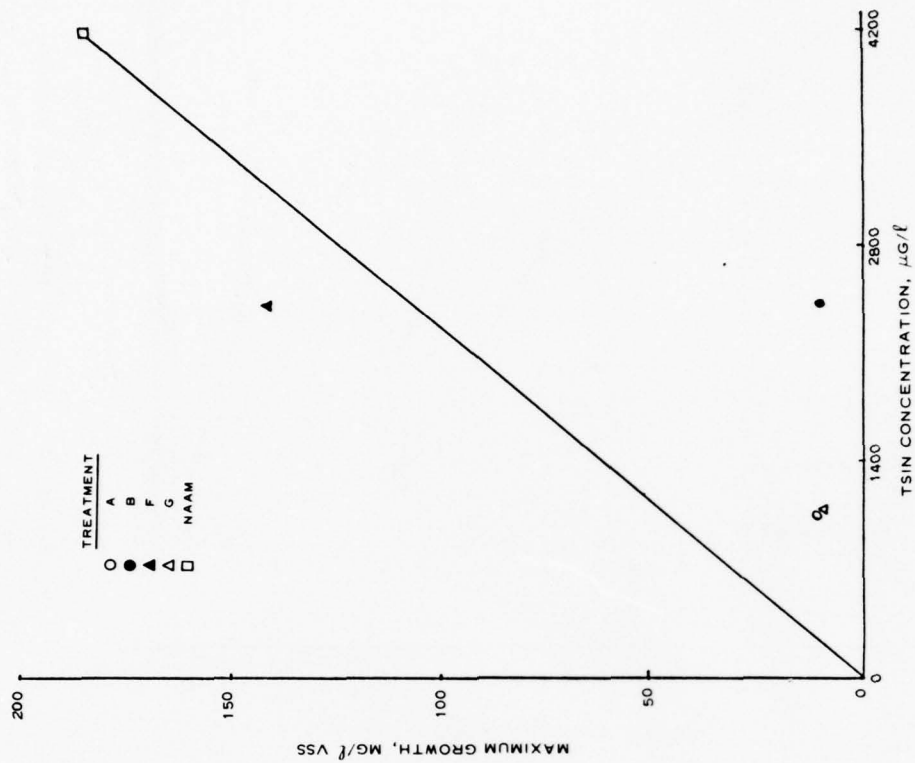


Figure A18. Maximum growth in sample GS800B (Run 1) as a function of TSIN

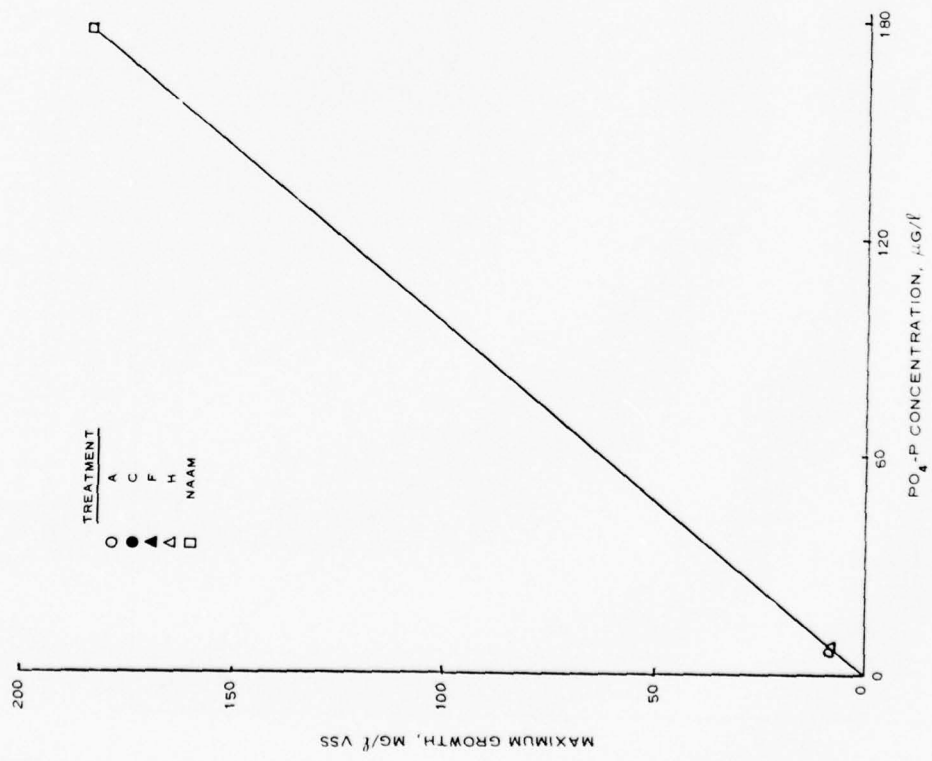


Figure A19. Maximum growth in sample GS800C (Run 1) as a function of PO₄-P

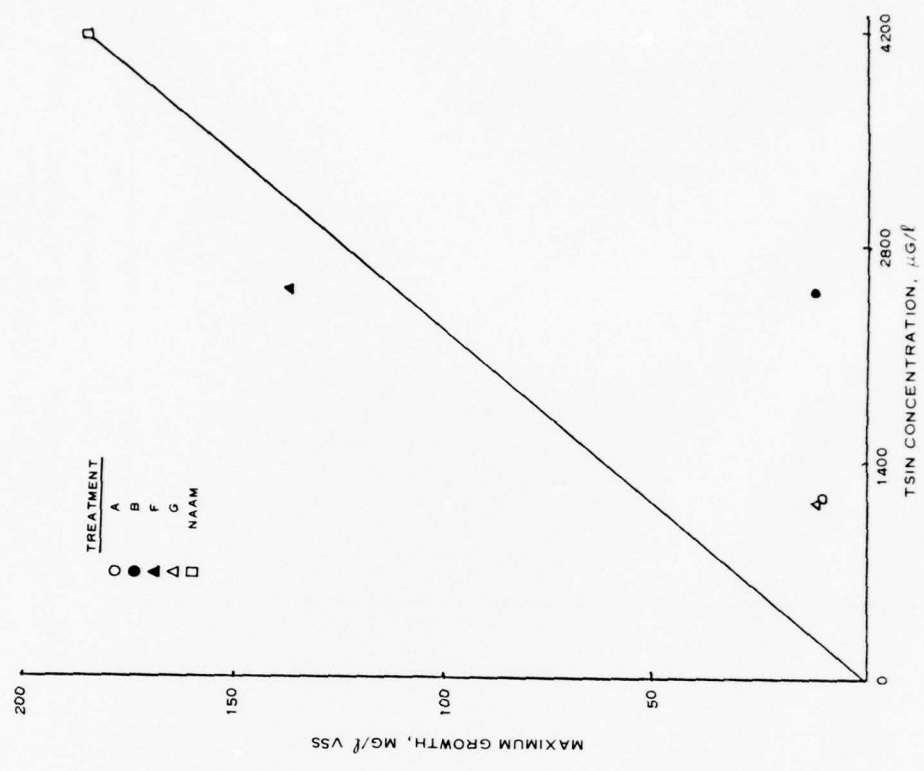


Figure A20. Maximum growth in sample GS800C (Run 1) as a function of TSIN

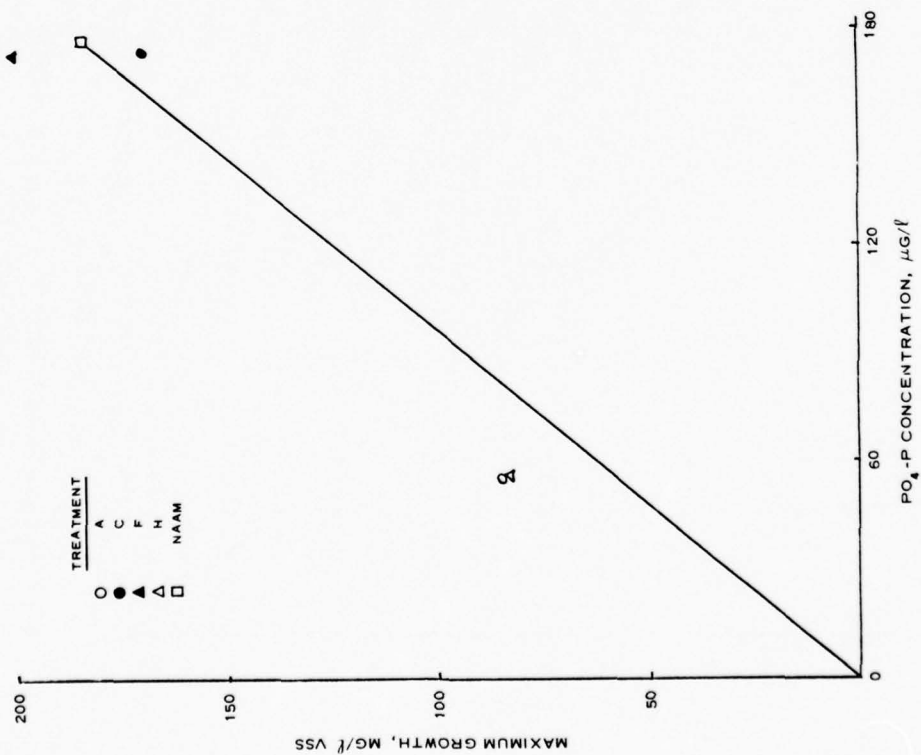


Figure A21. Maximum growth in sample GS770 (Run 1) as a function of $PO_4\text{-P}$

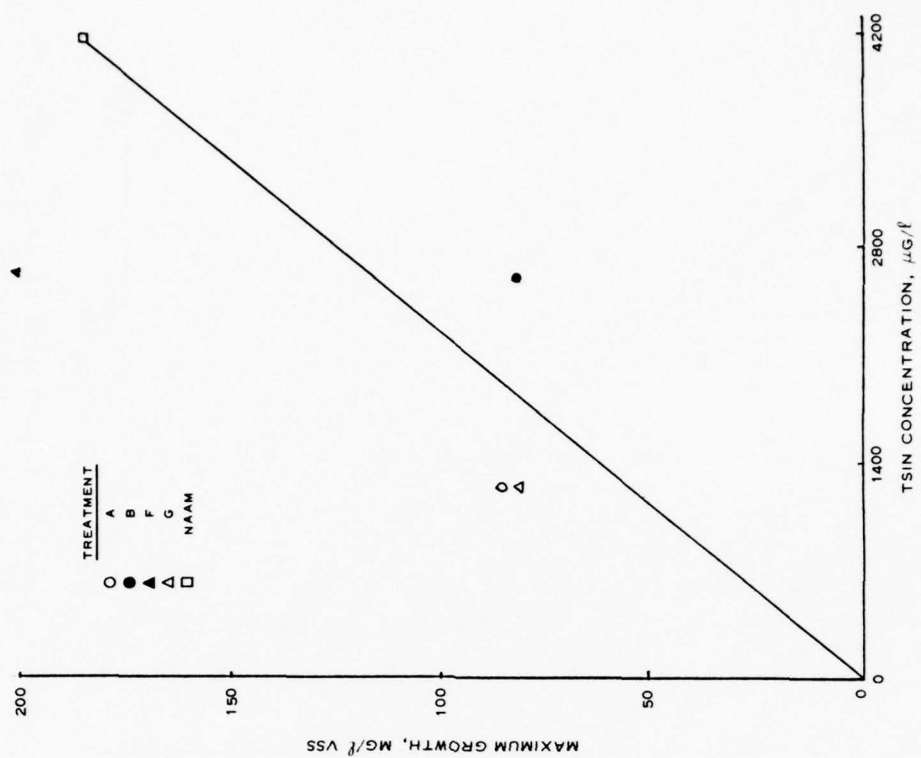


Figure A22. Maximum growth in sample GS770 (Run 1) as a function of TSIN

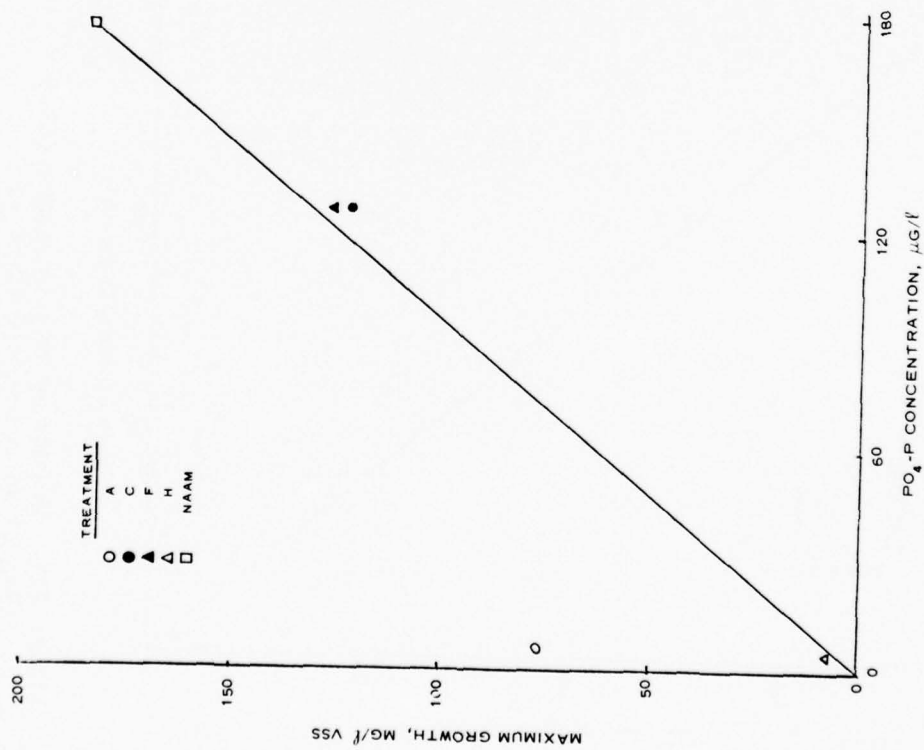


Figure A23. Maximum growth in sample GS738 (Run 1) as a function of $PO_4\text{-P}$

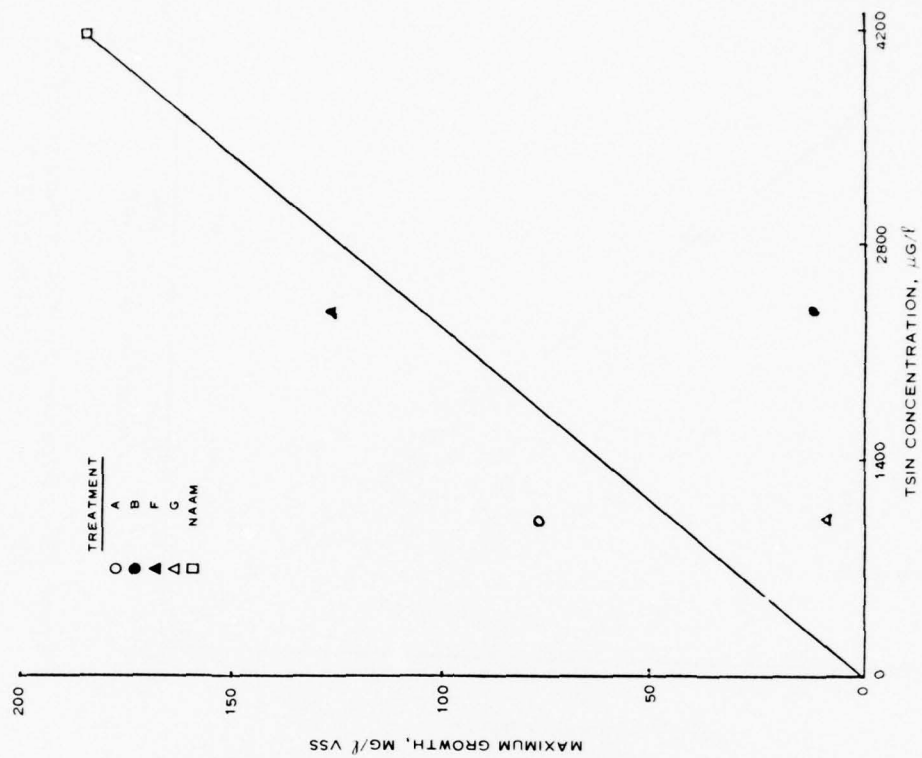


Figure A24. Maximum growth in sample GS738 (Run 1) as a function of TSIN

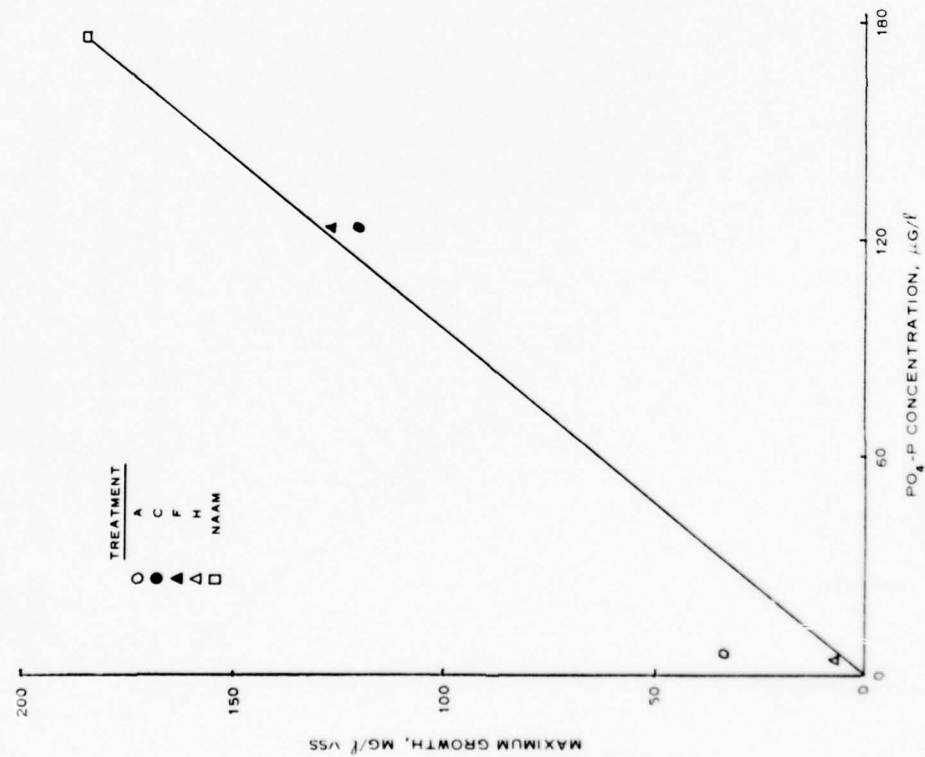


Figure A25. Maximum growth in sample GS700 (Run 1) as a function of $PO_4\text{-P}$

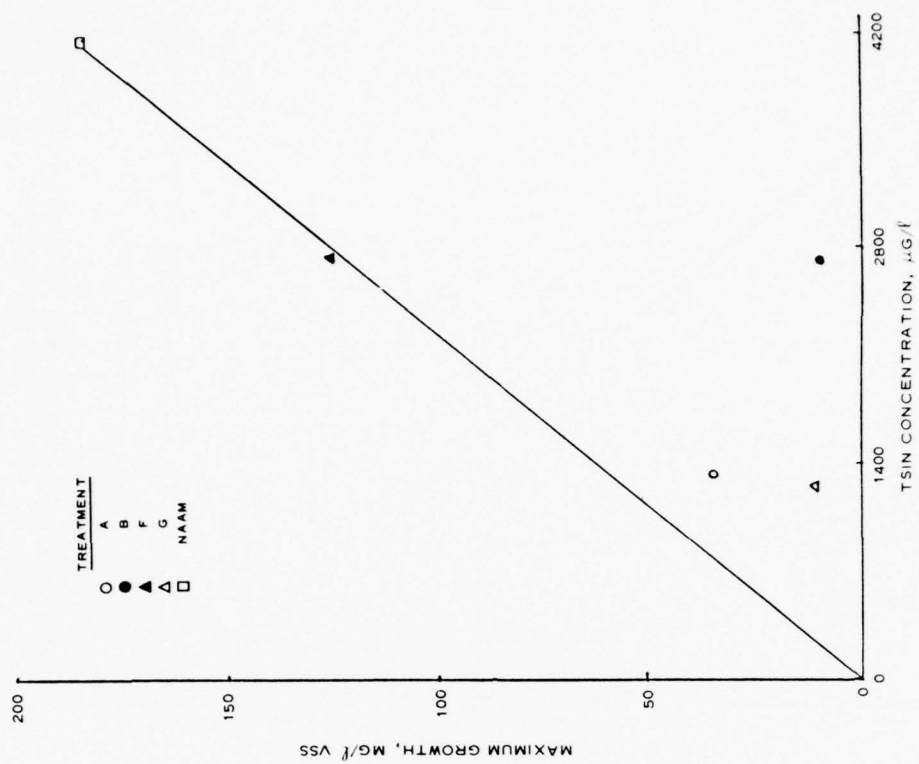


Figure A26. Maximum growth in sample GS700 (Run 1) as a function of TSIN

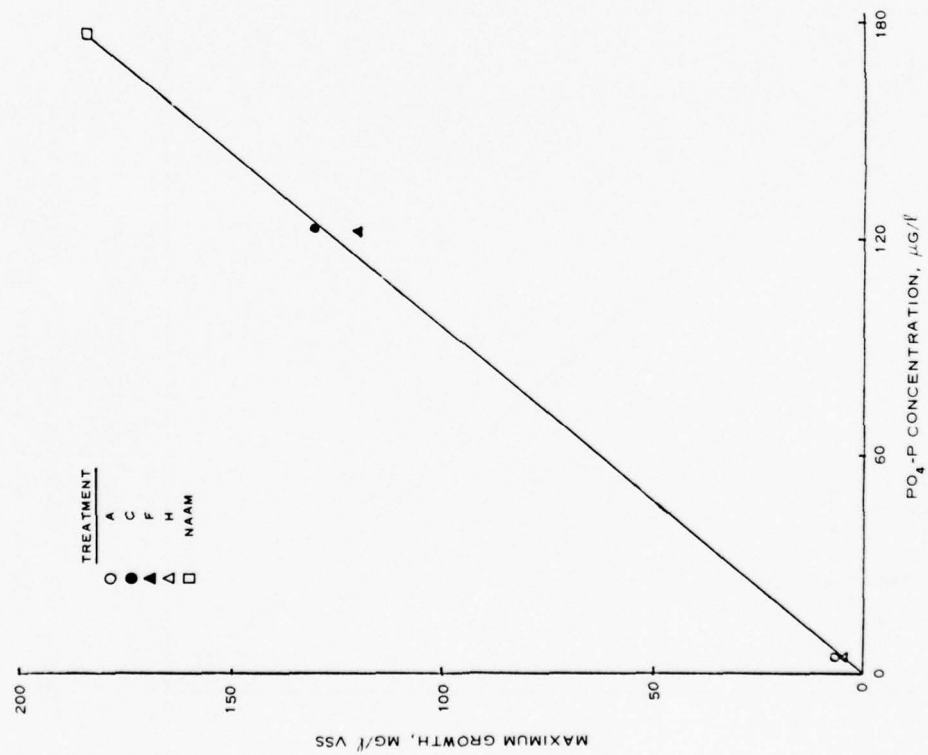


Figure A27. Maximum growth in sample GS695 (Run 1) as a function of PO₄-P

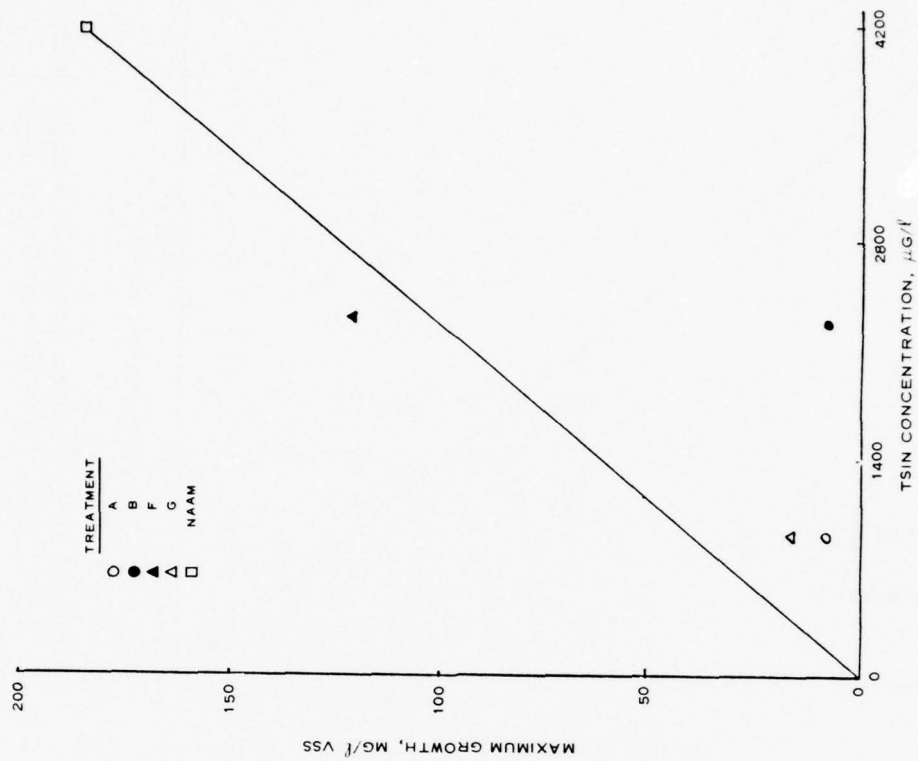


Figure A28. Maximum growth in sample GS695 (Run 1) as a function of TSIN

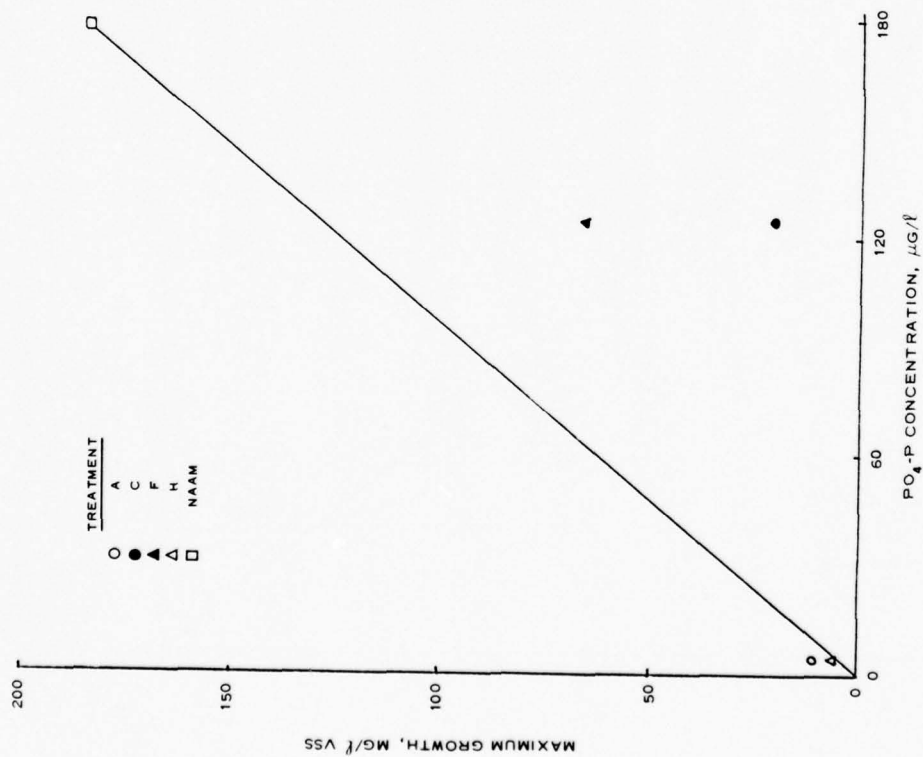


Figure A29. Maximum growth in sample Belt 1 (Run 1) as a function of $PO_4\text{-P}$

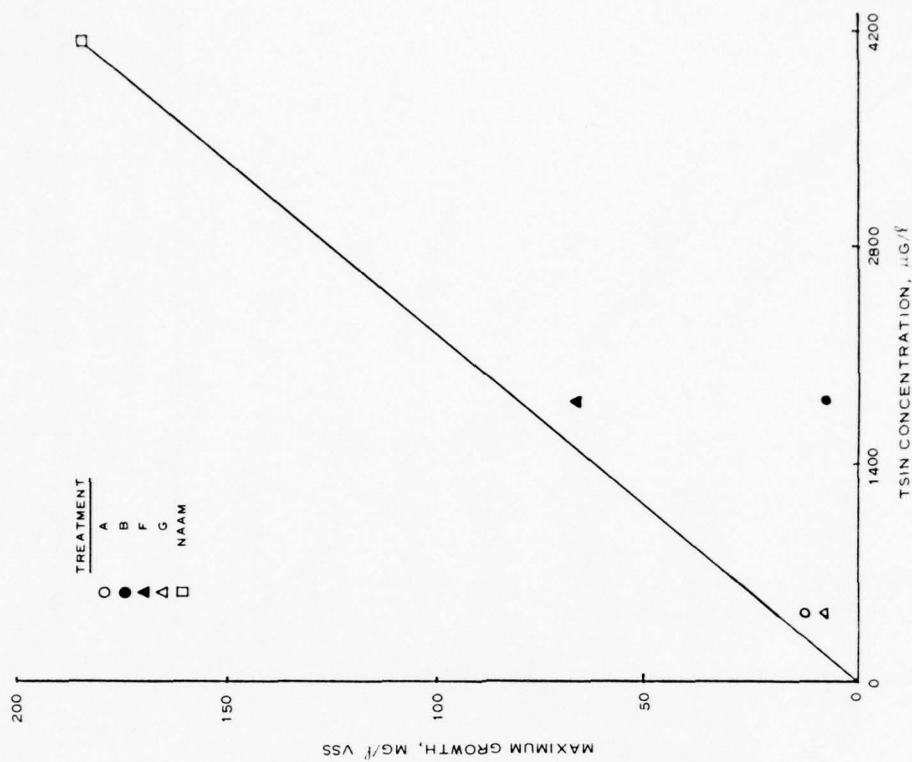


Figure A30. Maximum growth in sample Belt 1 (Run 1) as a function of TSIN

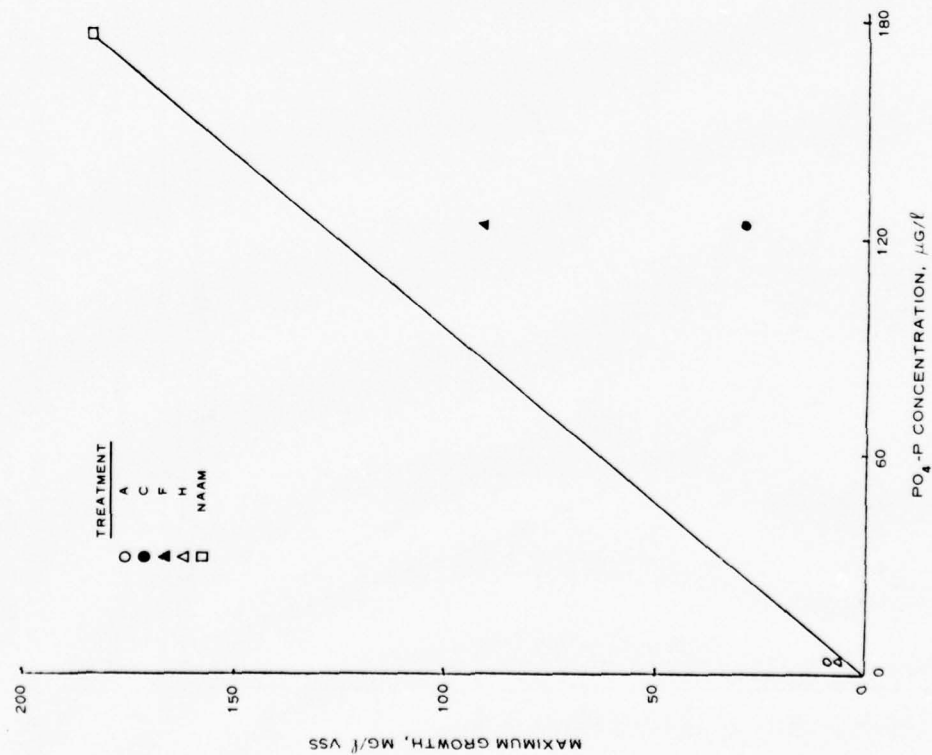


Figure A31. Maximum growth in sample Belt 2 (Run 1) as a function of PO_4-P

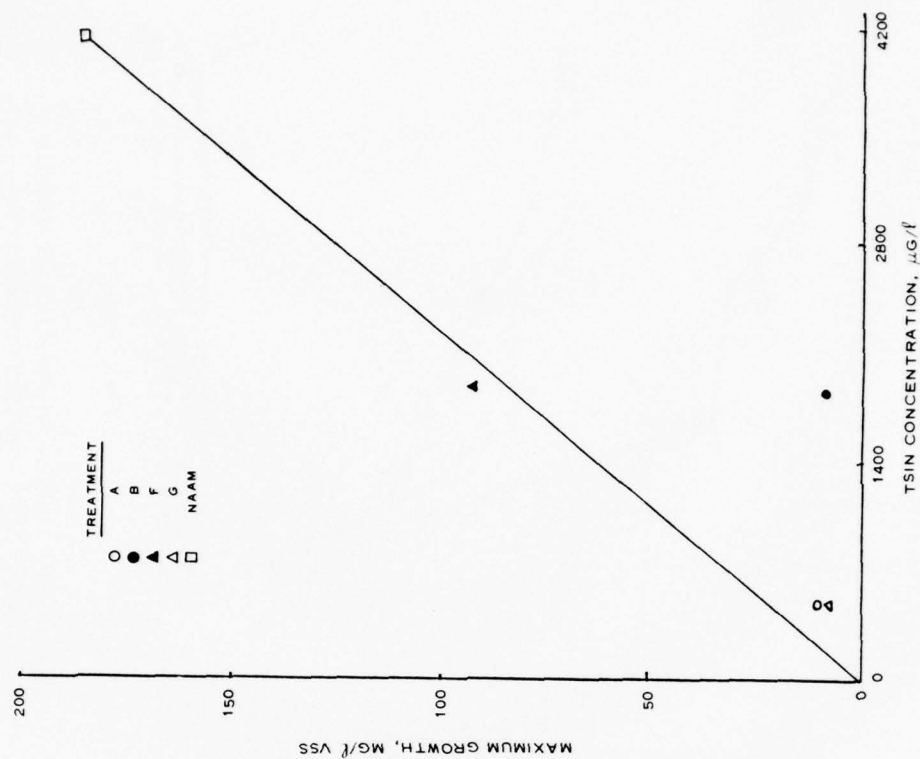


Figure A32. Maximum growth in sample Belt 2 (Run 1) as a function of TSIN

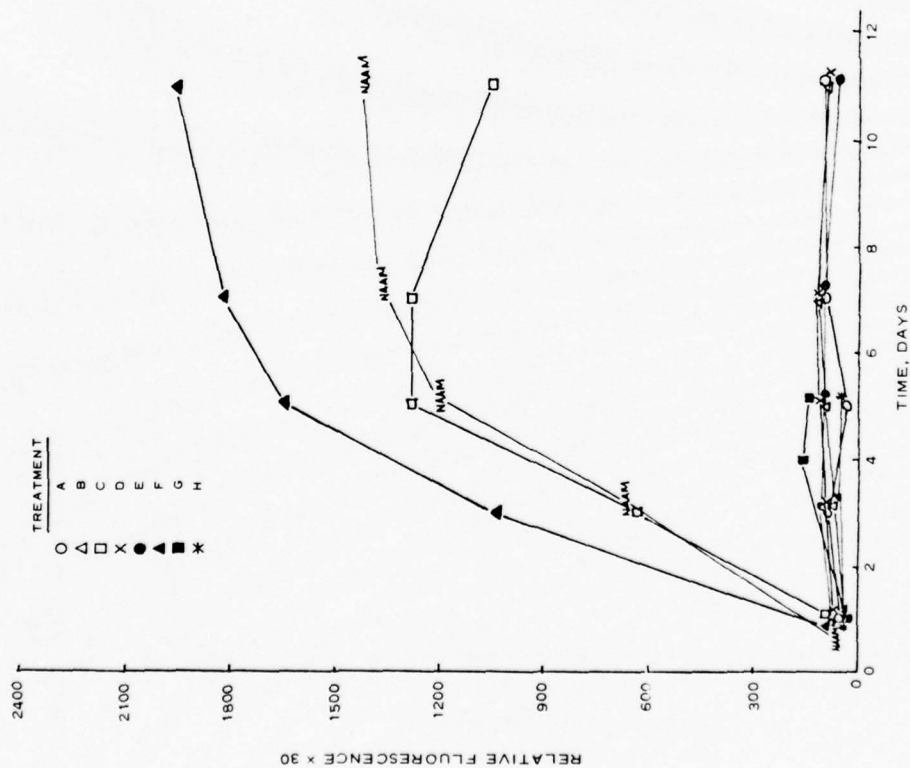


Figure A33. Relative fluorescence of sample GS800B with various treatments

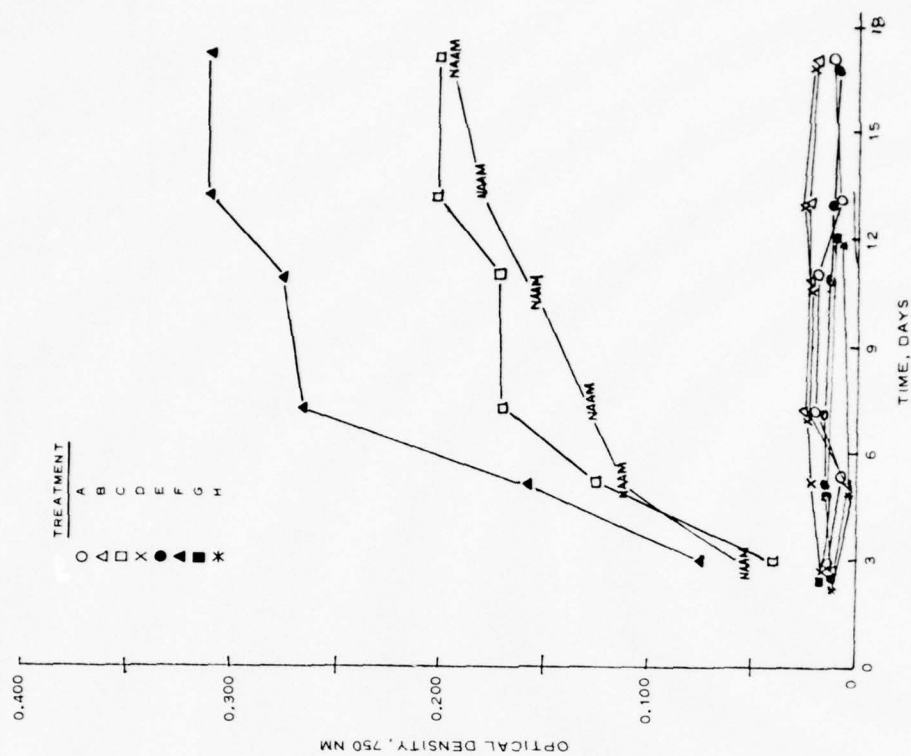


Figure A34. Optical density of sample GS800B with various treatments

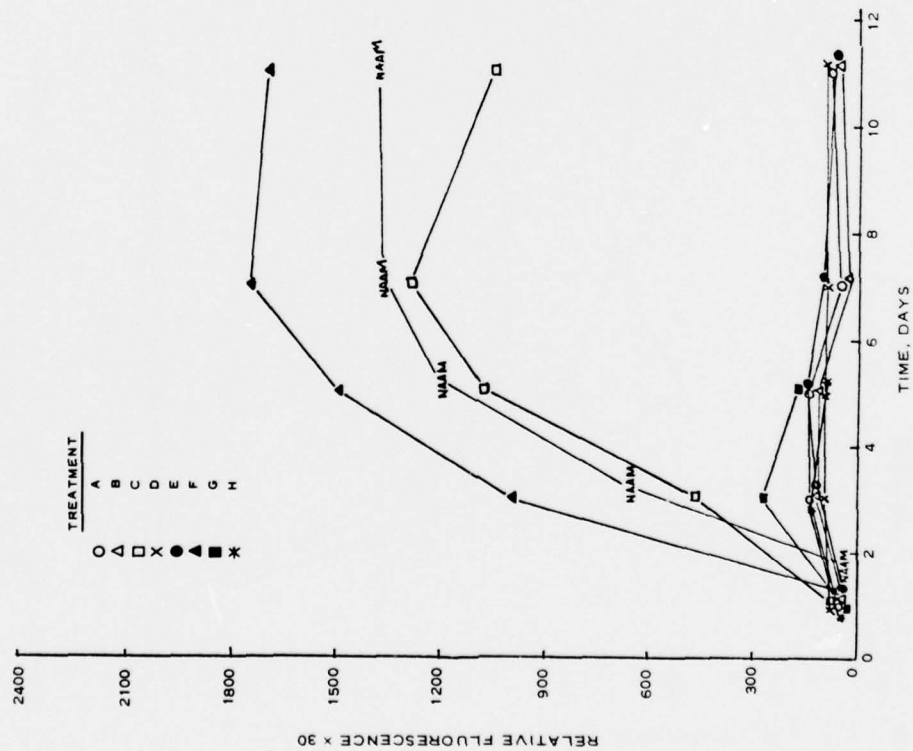


Figure A35. Relative fluorescence of sample GS800C with various treatments

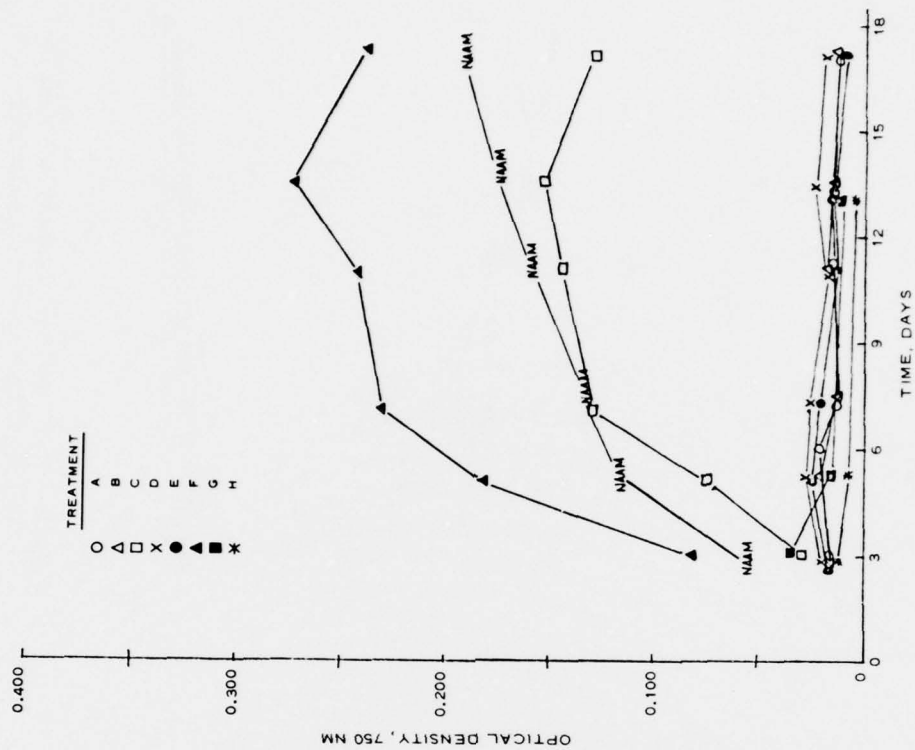


Figure A36. Optical density of sample GS800C with various treatments

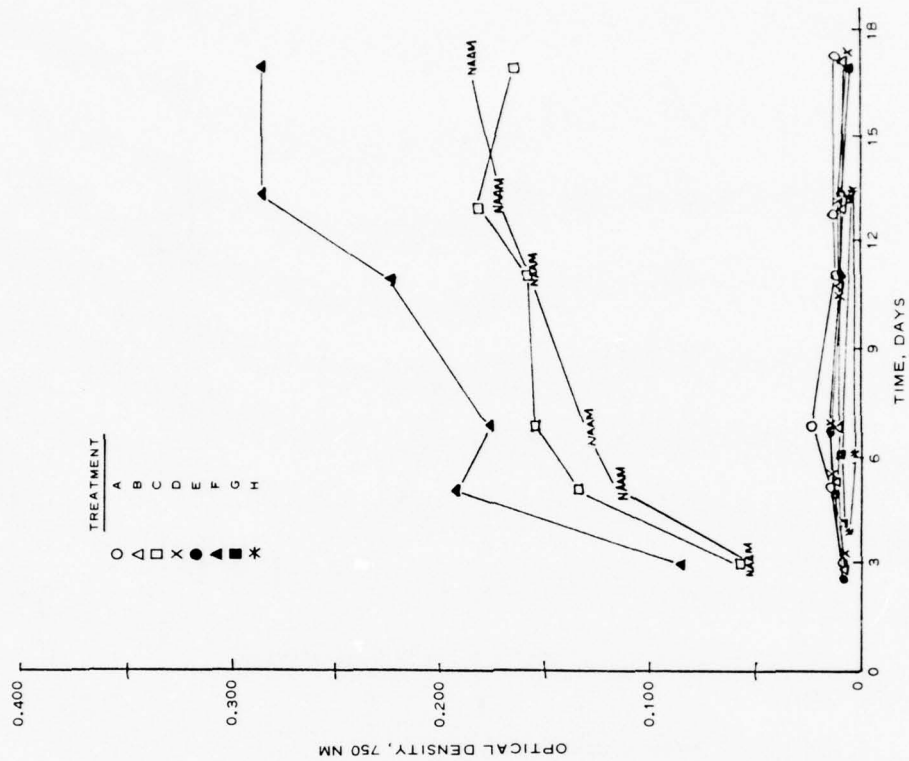


Figure A38. Optical density of sample GS738 with various treatments

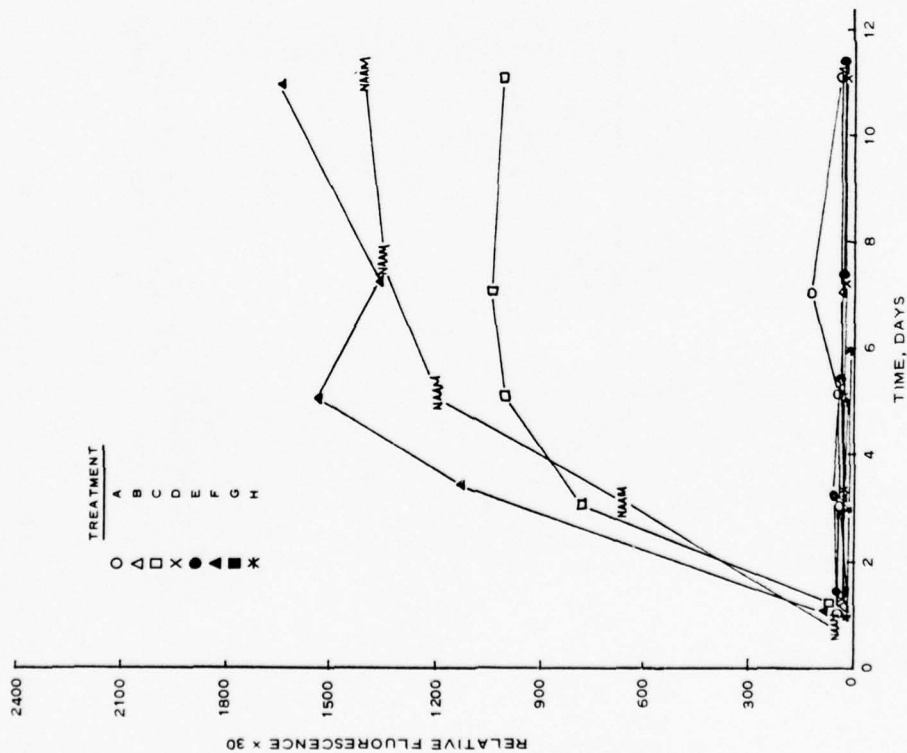


Figure A37. Relative fluorescence of sample GS738 with various treatments

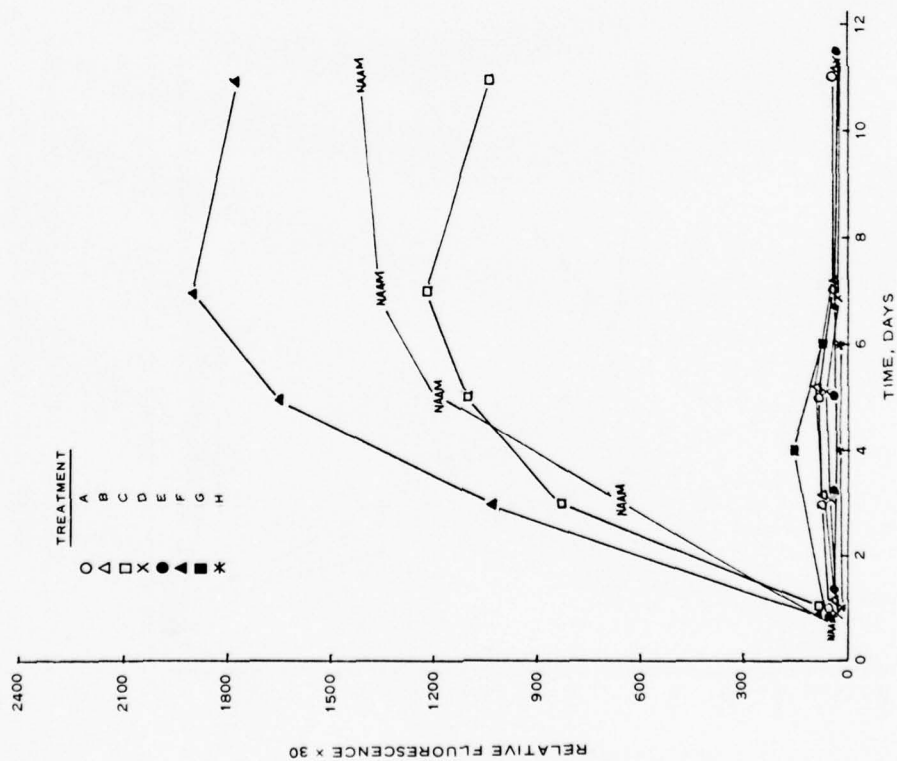


Figure A39. Relative fluorescence of sample GS700 with various treatments

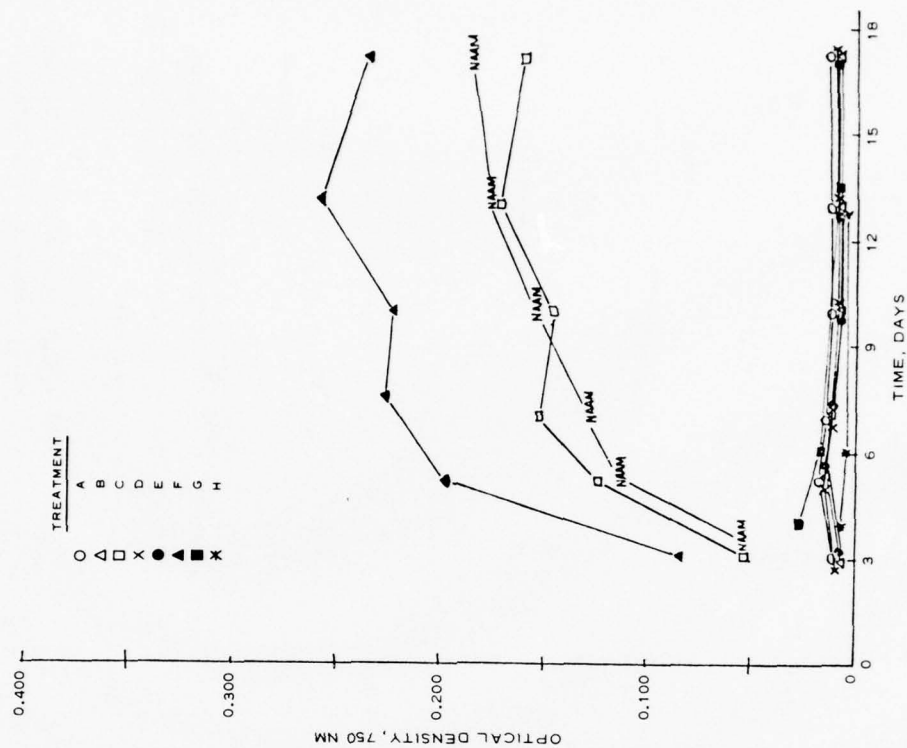


Figure A40. Optical density of sample GS700 with various treatments

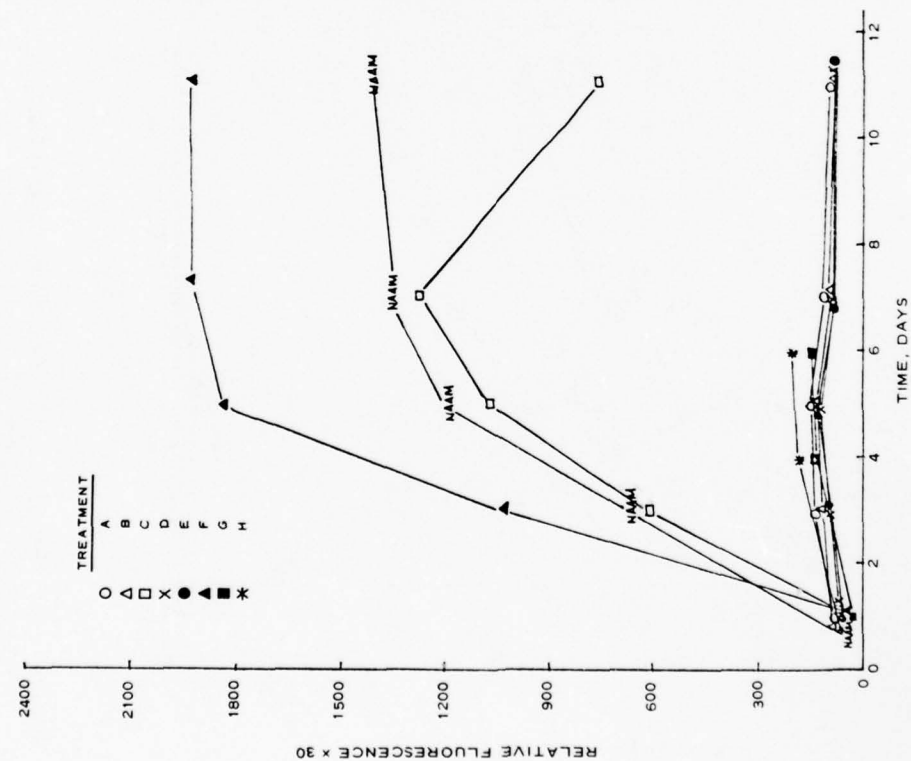


Figure A41. Relative fluorescence of sample GS695 with various treatments

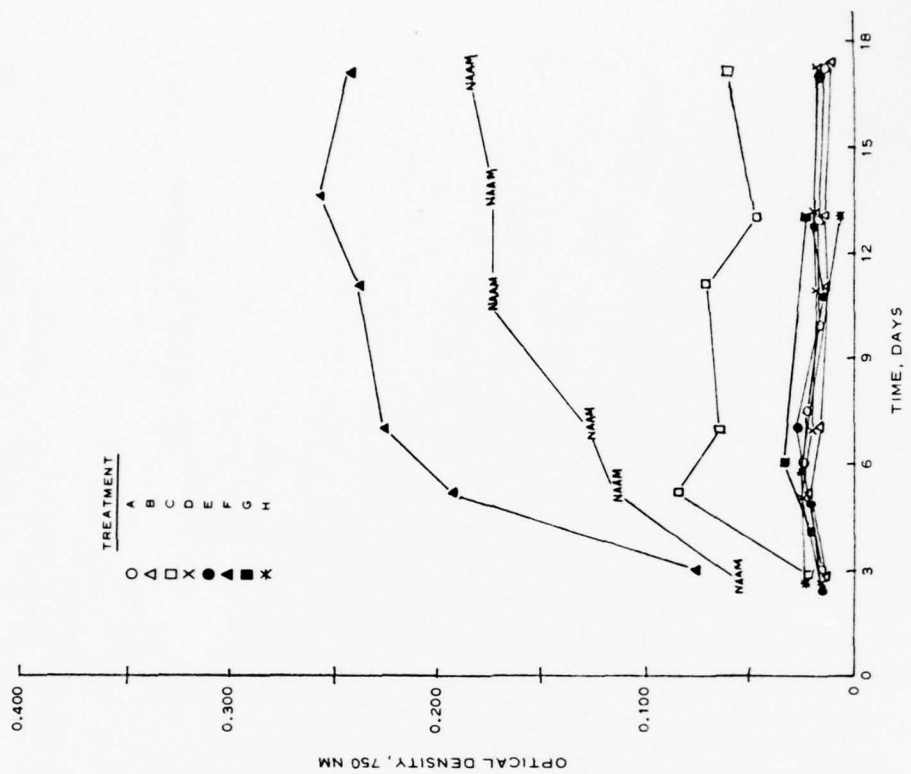


Figure A42. Optical density of sample GS695 with various treatments

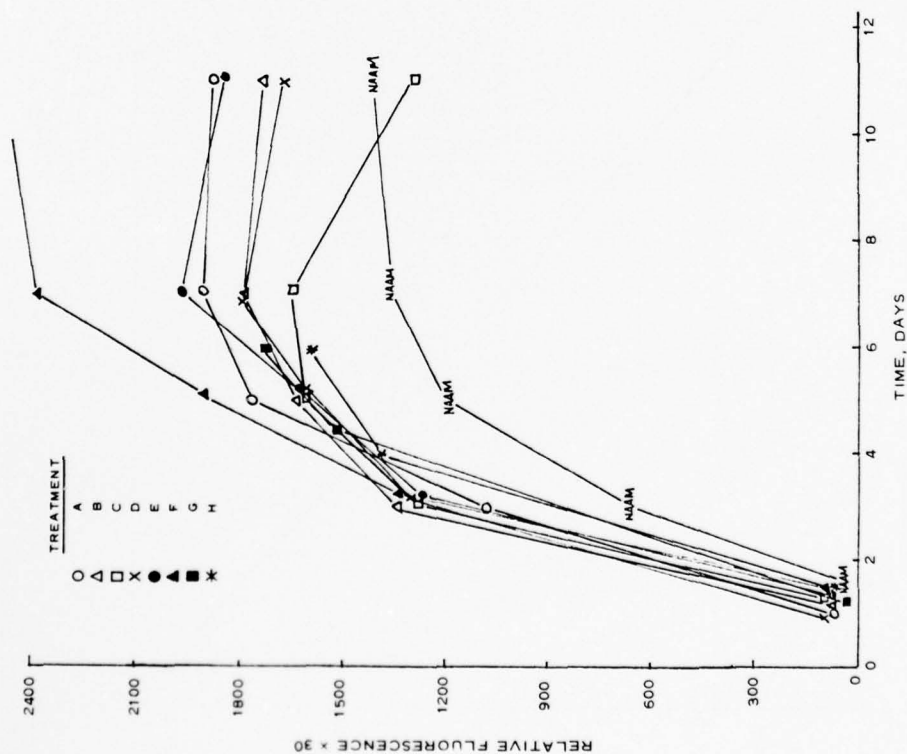


Figure A43. Relative fluorescence of sample GS770 with various treatments

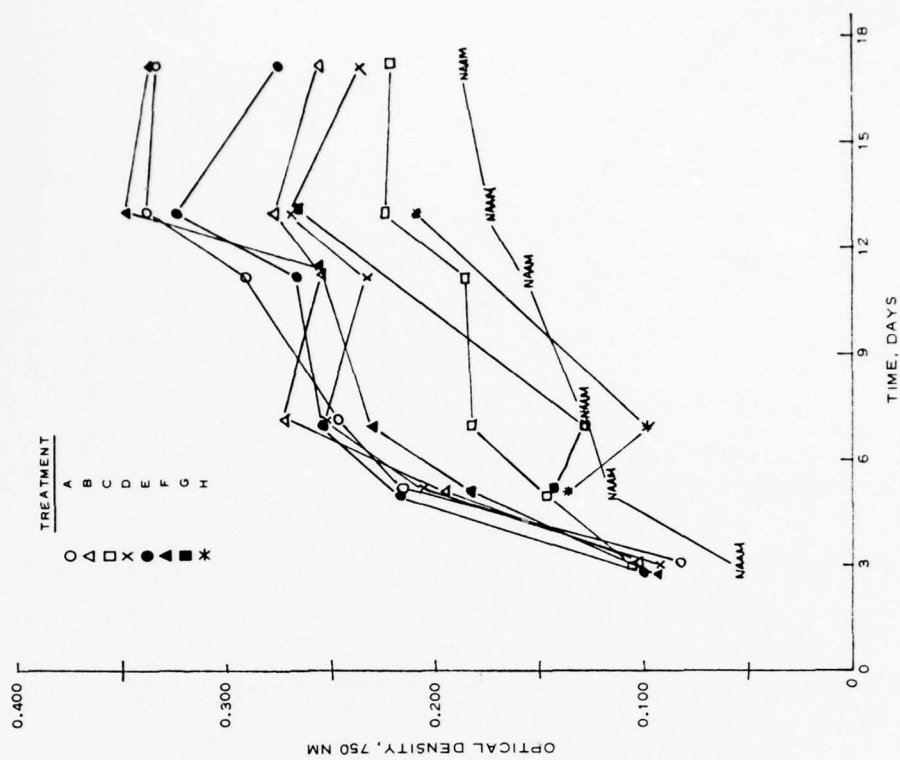


Figure A44. Optical density of sample GS770 with various treatments

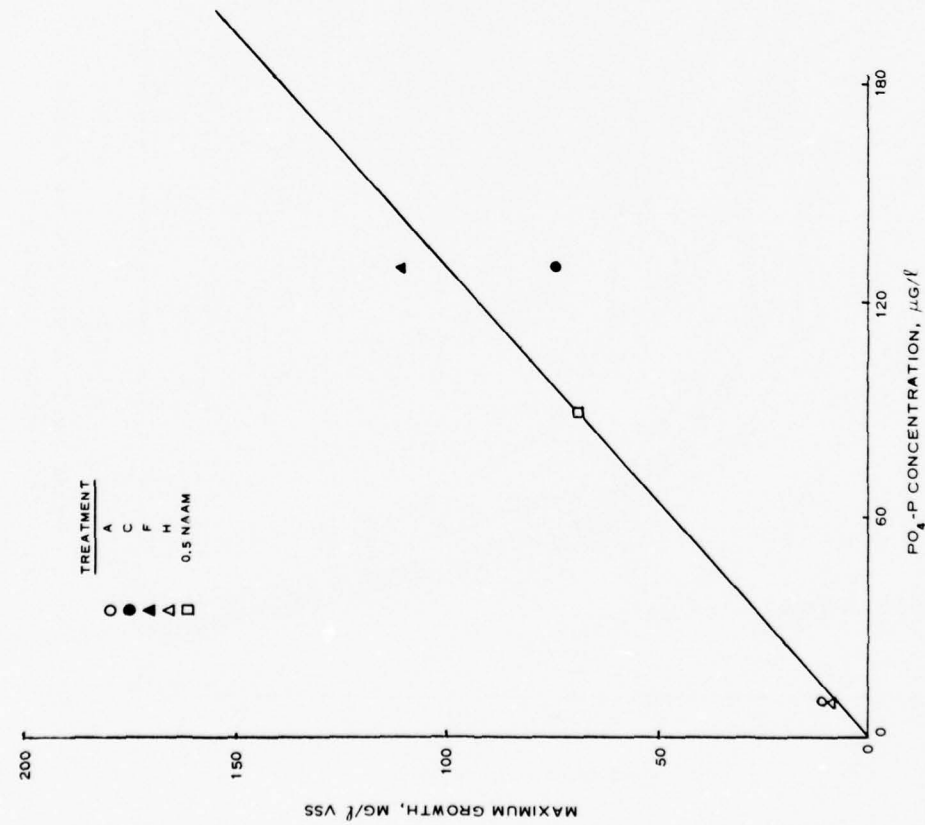


Figure A45. Maximum growth in sample GS800B (Run 2) as a function of $PO_4\text{-P}$

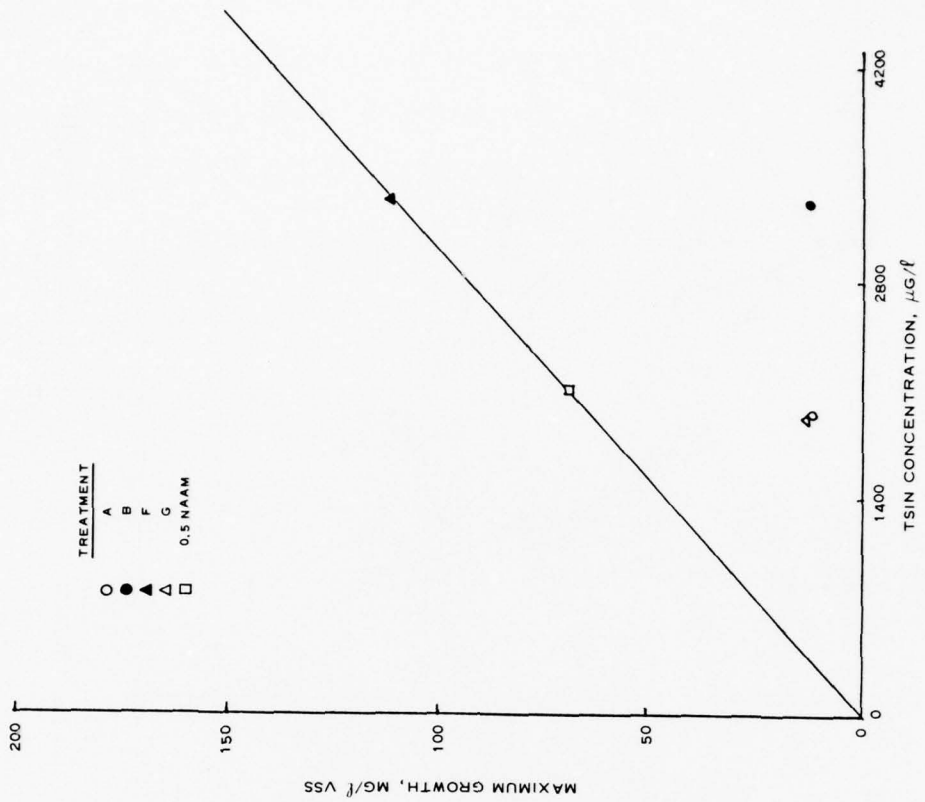


Figure A46. Maximum growth in sample GS800B (Run 2) as a function of TSIN

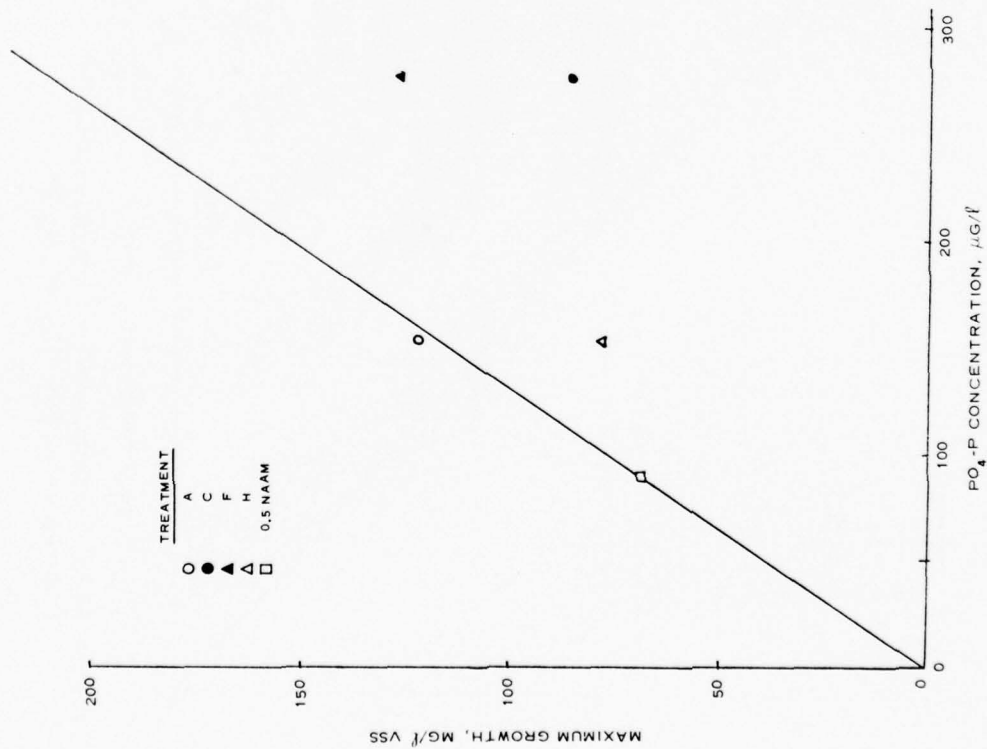


Figure A47. Maximum growth in sample GS770 (Run 2) as a function of $PO_4\text{-P}$

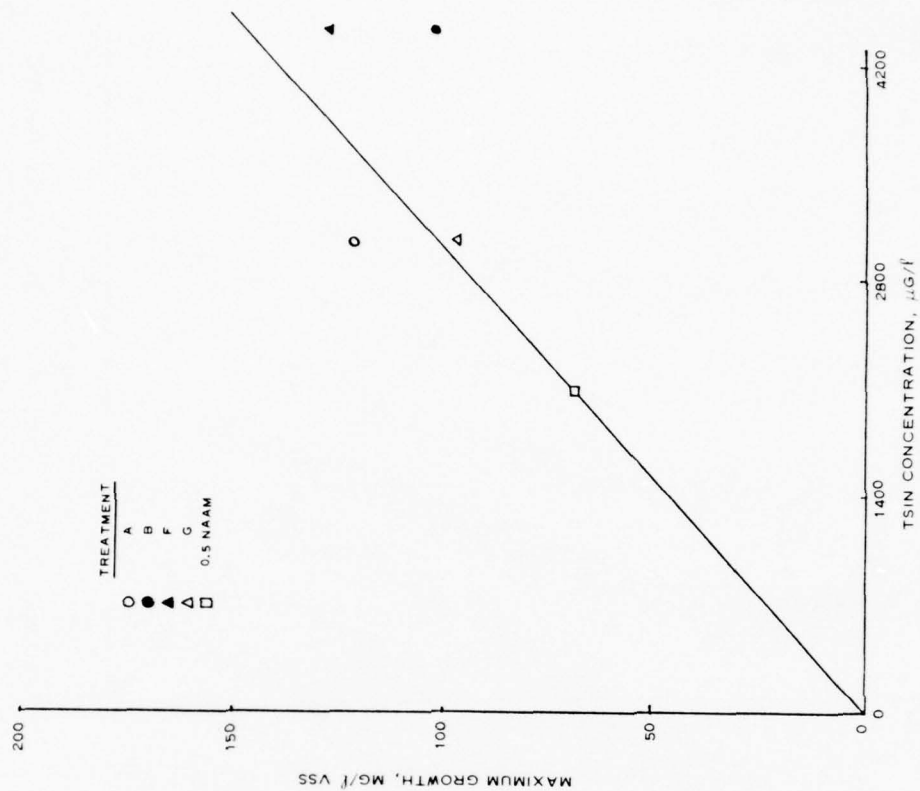


Figure A48. Maximum growth in sample GS770 (Run 2) as a function of TSIN

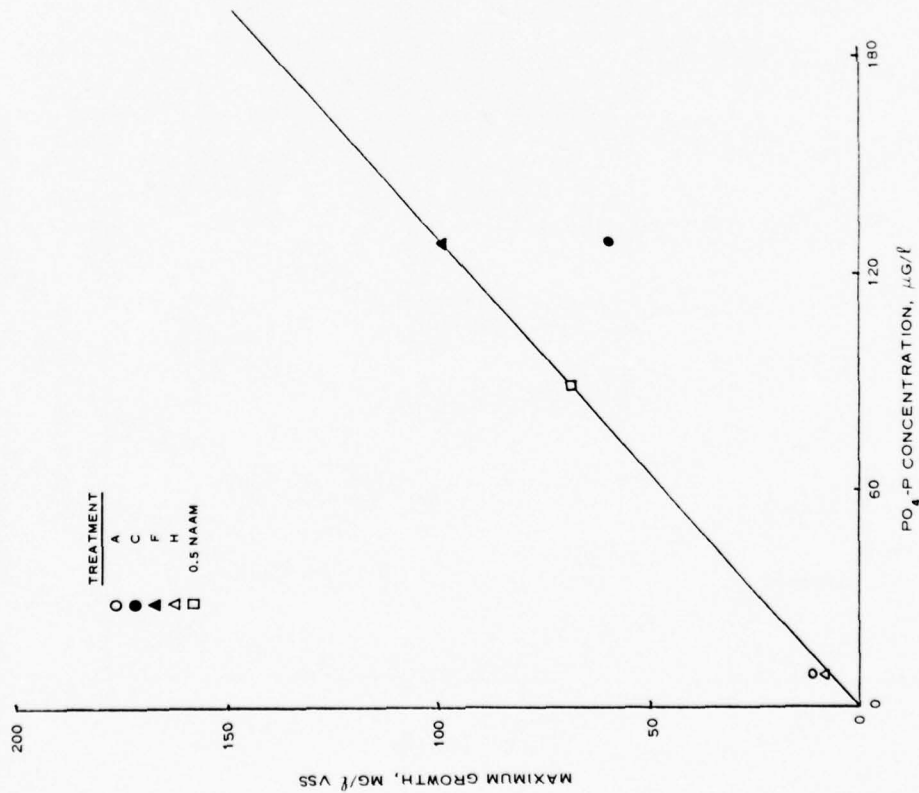


Figure A49. Maximum growth in sample GS800C (Run 2) as a function of PO₄-P

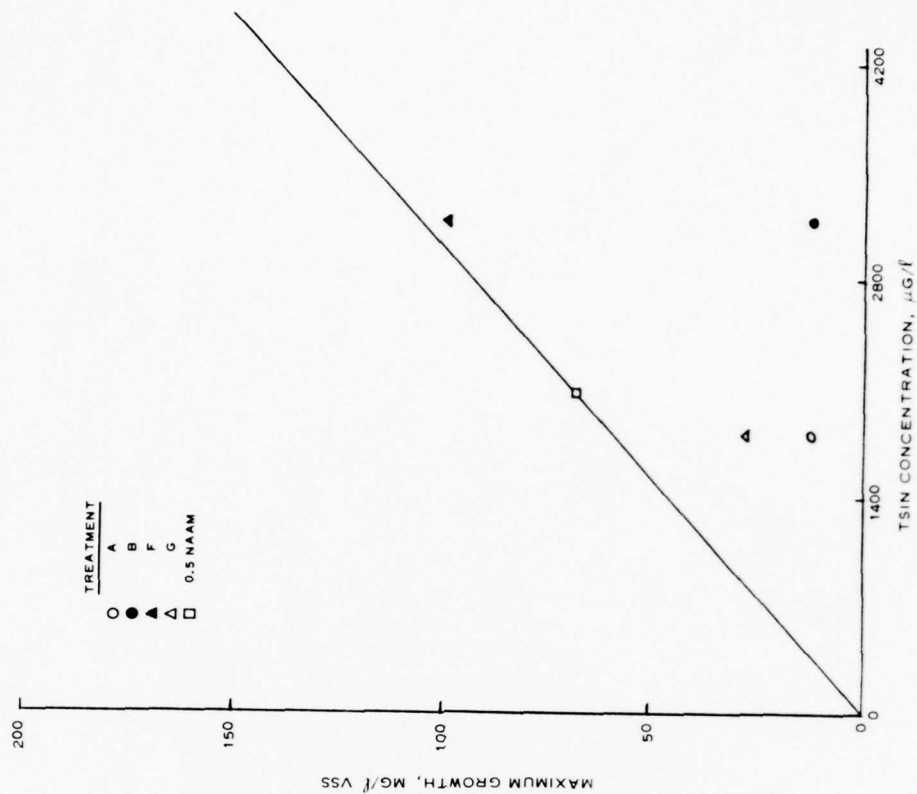


Figure A50. Maximum growth in sample GS800C (Run 2) as a function of TSIN

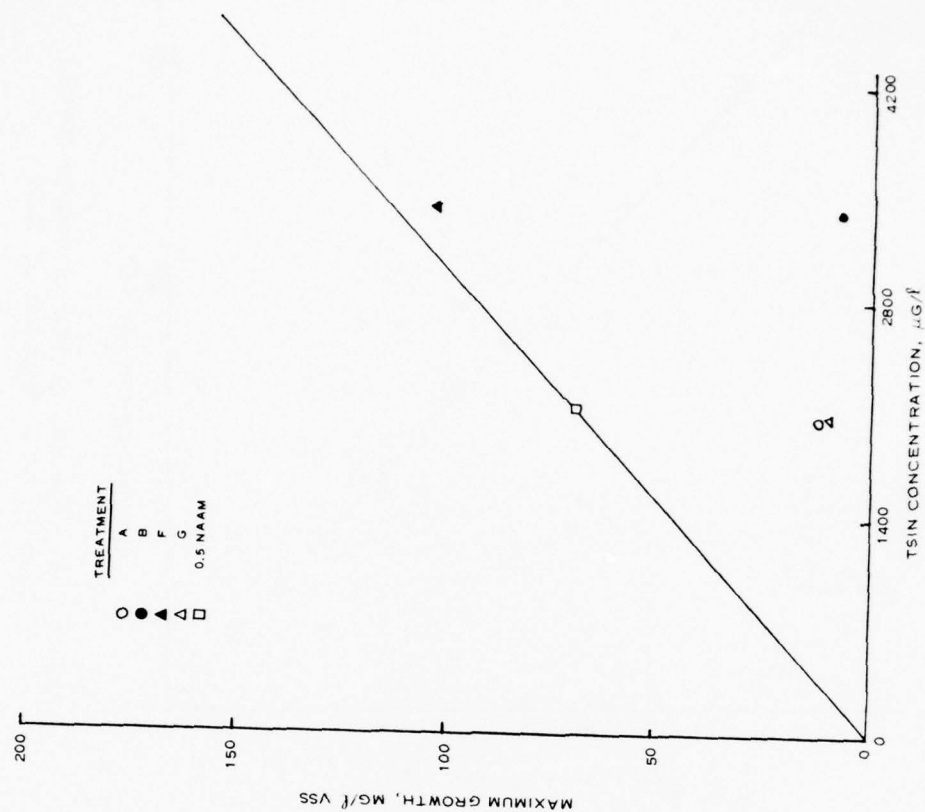


Figure A52. Maximum growth in sample GS738 (Run 2) as a function of TSIN

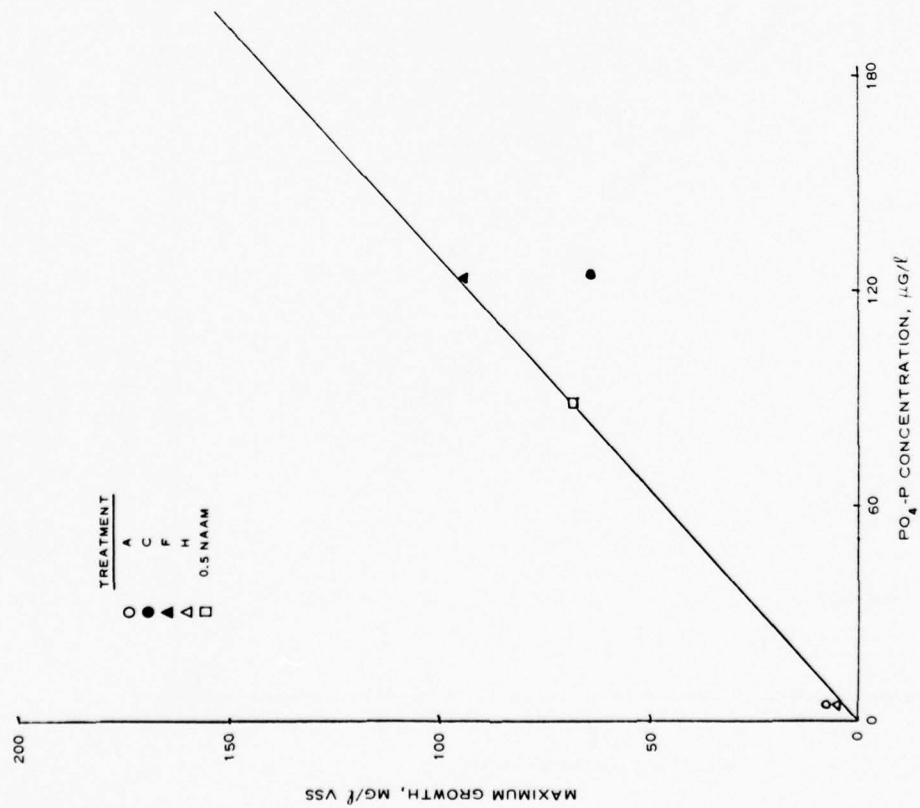


Figure A53. Maximum growth in sample GS700 (Run 2) as a function of $PO_4\text{-P}$

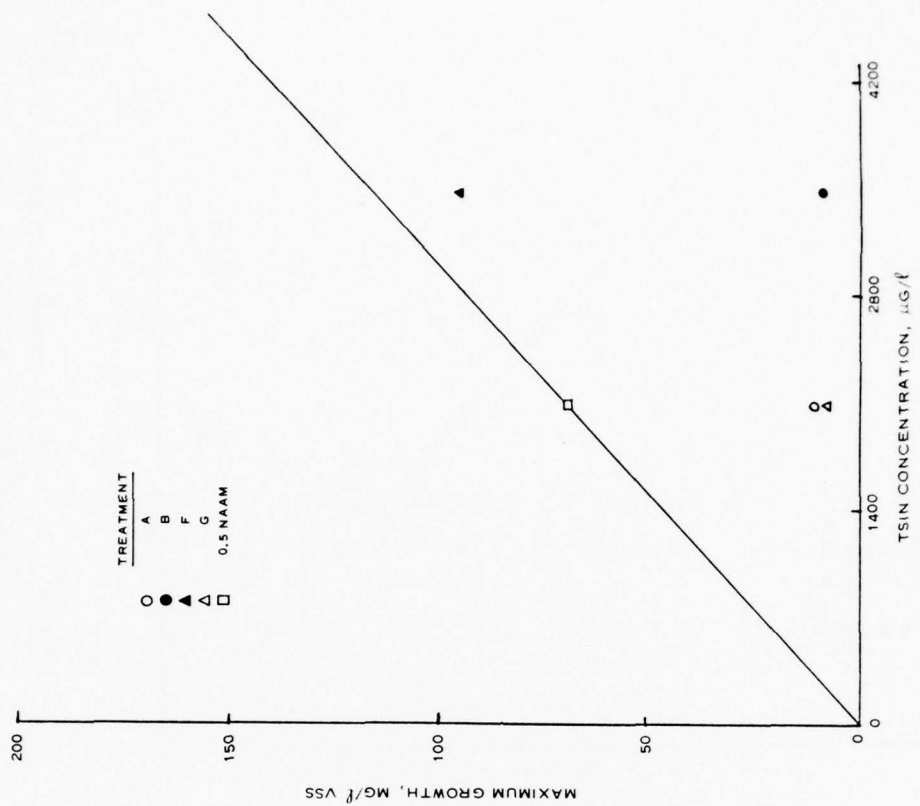


Figure A54. Maximum growth in sample GS700 (Run 2) as a function of TSIN

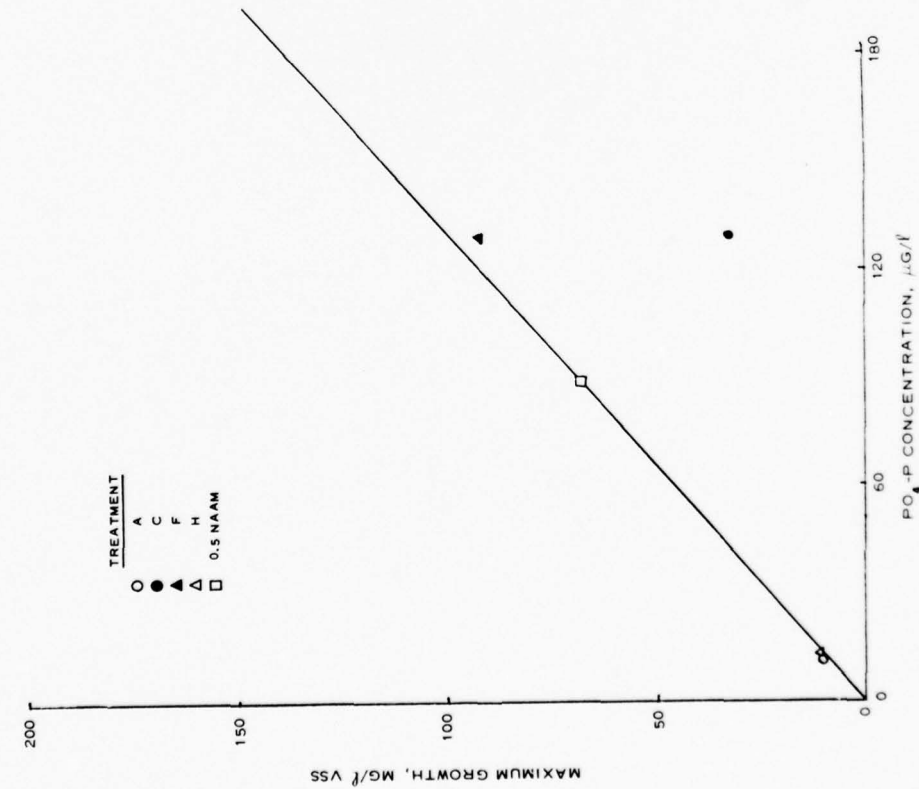


Figure A55. Maximum growth in sample GS695 (Run 2) as a function of PO₄-P

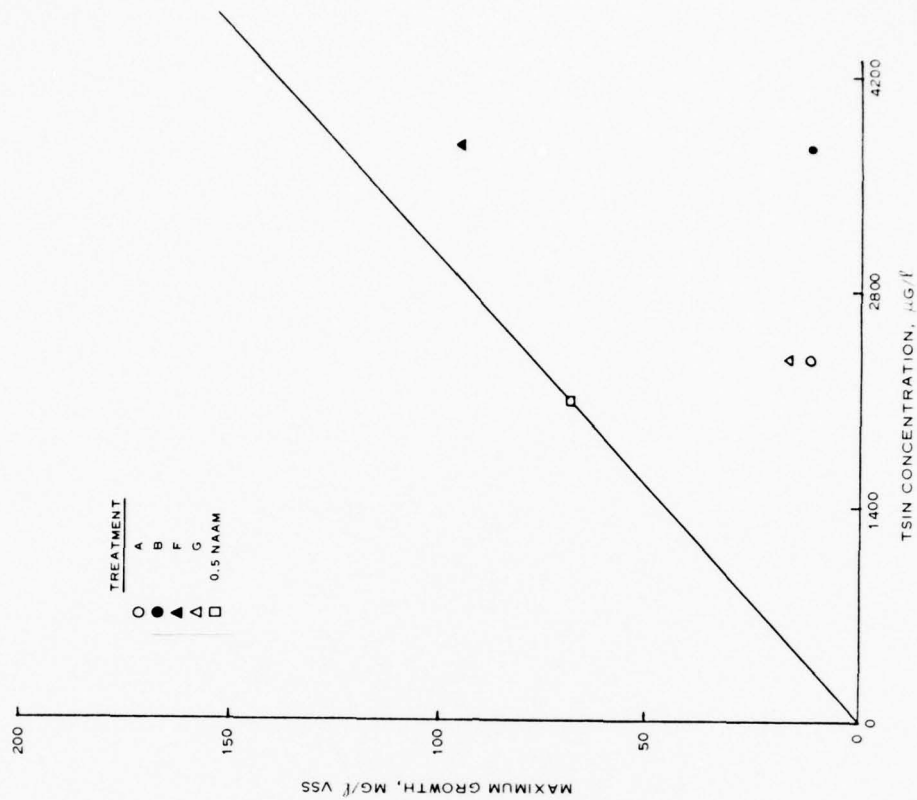


Figure A56. Maximum growth in sample GS695 (Run 2) as a function of TSIN

APPENDIX B: HYDROCOMP SIMULATION OF
JORDAN CREEK DRAINAGE BASIN

1. The hydrologic character of the Jordan Creek drainage basin was simulated using the Hydrocomp Simulation Program (HSP).^{*} The HSP model was fitted to the Jordan Creek data at the Schnecksville (Trexler damsite) and Allentown streamflow gages by calibration against the record of flow obtained from the U. S. Geological Survey (USGS) for the period of 1966-1976. The HSP permitted output of the streamflow at designated inflow locations to the proposed impoundment. Because more than 200 storm events occurred within the historic period, the effect of the proposed impoundment is indicated for a wide variety of storm situations.

Description of the Area

Physiography

2. The Jordan Creek watershed is situated in southeastern Pennsylvania near Allentown at about latitude 40°39' and longitude 75°38'. The drainage area is 77.8 square miles** at the Allentown stream gage (Figure B1). The watershed is roughly 15 miles across from east to west and 9 miles across from north to south and has rolling hills with mild slopes from 2 to 4 percent over most of its area. The elevation range is from 1450 to 293 ft with a common elevation of 500 ft. Jordan Creek flows about 28 miles to its confluence with the Lehigh River (Figure B2).

Hydrometeorological data

3. Figure B1 shows the locations of precipitation, temperature, wind, evaporation, radiation, and streamflow stations near and within the Jordan Creek watershed. Table B1 summarizes the available data.

4. In the calibration of the watershed response, the following data series were required: hourly precipitation, daily potential evapotranspiration, daily maximum and minimum temperature, daily wind

* Hydrocomp International, Inc. 1976. Hydrocomp Simulation Programming Operations Manual, Palo Alto, Calif.

** U. S. customary units are used throughout this appendix because the model simulations were made in terms of these units. A table of factors for converting these units to metric (SI) units is found on page B13.

movement, and solar radiation. Mean daily streamflow and storm hydrographs are necessary for the calibration.

5. The hourly precipitation gage at Allentown was used to represent the time distribution of rainfall over most (82 percent) of the basin. The Claussville and New Tripoli daily rain gage data were distributed into hourly values using the Allentown hourly data as a basis. No evaporation station exists in the watershed, and the Class A pan evaporation data at Landisville were used in calculating the potential evapotranspiration. Pan coefficients were taken from the U. S. Weather Bureau.* The temperature records at Allentown were found to be the most representative index of the temperature over the watershed area. Temperature, wind, and radiation data are used in the snowmelt calculations, specifically, melt-rate determinations, convection, and condensation melt.

6. Jordan Creek watershed is better gaged for streamflow than for any of the other hydrometeorological data series. The following tabulation lists the streamflow gages in the Jordan Creek watershed used in this study. The necessary stage hydrographs, rating curves, and streamflow records were obtained from the USGS District Office in Harrisburg, Pennsylvania.

<u>Name</u>	<u>USGS Gage No.</u>	<u>Drainage Area square miles</u>	<u>Period of Record</u>
Allentown	01452000	77.8	1946-Present
Schnecksville	01451800	52.3	1966-Present

7. The annual rainfall isohyets shown in Figure B3 were determined for use as an overlay. Since orographic lifting in the northwest area of the basin is the principal control on the precipitation, the isohyets follow the ridge lines.

Soils

8. In general, the dominant soil associations in the drainage

* U. S. Department of Commerce. 1959. Evaporation Maps for the United States. Technical Paper No. 37. Weather Bureau. Washington, D. C.

basin, as shown in Figure B4, are the Trexler, Montevallo-Trexler, Ryder-Duffield, and Washington-Duffield.* The proposed impoundment would lie predominantly within the Montevallo-Trexler soil associations in the central portion of the drainage basin.

9. The Trexler associations are characterized as moderately deep (about 3 ft), moderately acid (pH of 5.0-6.0), well drained, and moderately permeable. These silt loam soils (shale fragments) are common on the rolling hills and ridges. They are dark brown to yellow-brown in color, and they lay over glacial till weathered from shale and sandstone. Some areas are characteristically gravel with thin layers of silt loam on the surface.

10. The general description of the Montevallo-Trexler soil association indicates that the soils are shallow (about 2 ft), slightly acid (pH of 6-6.5), and well drained. These soils are dark grey-brown to yellow-brown in color and are gravelly silt loams in texture. They lay over frost-churned material weathered from shale and contain appreciable gravel at or near the surface.

11. The Ryder-Duffield soil associations are found in only a small area on the southern tip of the watershed. They are deep (about 5 ft), well drained, near neutral reaction (pH of 7.0), and loamy. They have a dark brown to yellowish-brown surface soil that lies over calcareous, shaly limestone.

12. The Washington-Duffield soil associations occupy the lower portion of the Jordan Creek watershed. They are moderately deep (about 4 ft), slightly acid (pH of 6.5), well drained, and moderately permeable. These silt loam soils are dark brown to yellow-red-brown in color and lie on top of glacial till weathered from limestone.

Hydrocomp Simulation Program

13. The HSP is a computer program that is capable of modeling the hydrologic response of a watershed to input precipitation and

* U. S. Department of Agriculture. 1963. Soil Survey. Lehigh County, Pennsylvania. Soil Conservation Service. Series 1959, No. 31. Washington, D. C.

evaporation. The majority of the parameters are defined by the available hydrologic data, topographic maps, and geologic information. A field survey of the dimensions of the stream channels was accomplished to determine the hydraulic characteristics of Jordan Creek.

14. The Jordan Creek basin was partitioned into six segments* in relation to the input of the proposed impoundment. In addition, 15 channel reaches** were specified that control the streamflow routing. A reach point and segment were designated for each of the seven inlets to the proposed impoundment as shown in Figure B5. The characteristics for the 6 segments and 15 reaches are shown in Tables B2 and B3.

15. The calibration of the Jordan Creek watershed was established through several computer runs. The calibration parameters were set using only the LANDS module. No fine tuning of the parameters was accomplished using the CHANNEL module. Also, at the time of calibration, the snowmelt relations were estimated from the temperature data as the wind, radiation, and snow-depth information were not on-line at this time. Consequently, the hydrographs were calibrated only for the months of April through October using the 10-year data set for all segments.

16. The calibration parameters appear in Table B4. The parameters determined by the calibration process were LZSN, INFILTRATION, INTERFLOW, and KK24. The simulated and recorded annual flow volumes for the calibration period of April through October are shown in Table B5. The simulated flows are generally in good agreement with the recorded values.

17. Figure B6 shows the simulated storm hydrograph for a storm in August 1969. The good reproduction of the hydrographs indicates that the calibration could be only slightly improved by adjustment of the hourly output of the CHANNEL module. Therefore, the parameters developed were used for the discharge simulation as input of the proposed impoundment.

* A segment is a land area having uniform rainfall.

** A reach is a length of stream channel that has uniform hydraulic characteristics.

Table B1
Hydrometeorological Stations Used in the Study

<u>Name</u>	<u>No.</u>	<u>Gage Type</u>	<u>Period of Record, yr</u>
Allentown	0106	Hourly precipitation; daily max-min temperature	1948-Present
Palmerton	6689	Hourly precipitation; daily max-min temperature	1948-Present
Claussville	1505	Daily precipitation	1948-Present
New Tripoli	6326	Daily precipitation	1948-Present
Landisville	4778	Daily evaporation; wind	1952-Present
Washington, D. C.	0560	Daily radiation	1952-Present
Lehigh*	0634	Hourly precipitation	1948-Present
Leighton*	4934	Hourly precipitation	1948-Present
Philipsburg, N. J.*	6916	Hourly precipitation	1948-Present
Sellersville*	7938	Hourly precipitation	1948-Present
Tamac Dam*	8763	Hourly precipitation	1948-Present
Kresgeville*	4672	Daily precipitation	1948-Present
Landisville*	4778	Daily precipitation	1952-Present
Zionville*	9995	Daily precipitation	1950-Present

* Station used in isohyetal map development.

Table B2
Segment Characteristics for the HSP Model

<u>Segment</u>	<u>Area square mile</u>	<u>Percent Subwatershed Basin</u>		<u>Mean Elevation ft msl</u>	<u>Mean Rainfall in.</u>	<u>Assigned Rain Gage</u>
		<u>1</u>	<u>2</u>			
1	23.05	44.1	29.6	600	43.99	New Tripoli
2	8.44	16.1	10.8	625	44.19	Claussville
3	4.42	8.4	5.7	582	44.07	Palmerton
4	16.41	31.4	21.1	450	44.19	Claussville
5	12.44		16.0	520	44.19	Claussville
6	13.04		16.8	304	44.12	Allentown

Table B3
Mean Channel Dimensions for the 15 Reach Locations

Reach	Stream Length miles	Drainage Area square mile	Elevation ft msl		Channel Width, ft		Incised Channel Depth ft	Slope of the Floodplain	Manning's Roughness Coefficient, NN	
			Up	Down	Stream	Floodplain			Channel	Floodplain
4	3.4	7.87	820	585	15	150	3	0.010	0.05	0.08
8	3.4	6.74	585	504	40	200	3	0.010	0.05	0.08
11	3.0	5.29	800	650	10	200	2	0.020	0.05	0.08
12	3.0	3.15	650	504	15	100	2	0.010	0.05	0.08
16	1.4	0.86	660	504	5	75	1.5	0.025	0.05	0.08
20	1.9	1.33	710	504	6	75	2	0.020	0.05	0.08
24	2.1	3.28	750	504	10	50	4	0.030	0.05	0.08
28	2.6	2.97	670	504	6	50	2	0.030	0.05	0.08
30	3.2	4.42	660	504	15	200	3	0.010	0.05	0.08
32	7.1	16.41	504	395	50	100	3	0.035	0.05	0.08
36	4.1	5.02	395	350	168	200	8	0.005	0.05	0.08
38	2.5	4.13	690	518	12	100	3	0.015	0.05	0.08
40	3.4	3.29	518	345	20	300	3	0.020	0.05	0.08
44	3.4	5.88	350	319	64	200	5	0.015	0.05	0.08
48	6.9	7.16	319	258	70	600	7.5	0.005	0.05	0.08

Table B4
HSP Simulation Parameters, Jordan Creek Watershed

Parameter*	Segment					
	1	2	3	4	5	6
K1	0.993	0.965	0.993	0.961	0.99	0.98
A	0.0	0.0	0.0	0.0	0.0	0.0
EPXM	0.10	0.10	0.10	0.10	0.10	0.10
UZSN	2.30	2.30	2.41	1.96	2.42	1.75
LZSN	14.36	14.36	15.07	12.24	15.14	10.96
K3	0.4	0.4	0.4	0.4	0.4	0.4
K24L	0.0	0.0	0.0	0.0	0.0	0.0
K24EL	0.0	0.0	0.0	0.0	0.0	0.0
INFILTRATION	0.02	0.02	0.02	0.02	0.03	0.03
INTERFLOW	5.0	5.0	5.0	5.0	3.0	3.0
L	200	200	200	200	200	200
SS	0.05	0.05	0.05	0.05	0.05	0.05
NN	0.30	0.30	0.30	0.30	0.30	0.30
IRC	0.60	0.60	0.60	0.60	0.60	0.60
KV	1.0	1.0	1.0	1.0	1.0	1.0
KK24	0.98	0.98	0.98	0.98	0.98	0.98

* K1 = Ratio of average segment rainfall to average gage rainfall.
 A = Impervious fraction of watershed surface area.
 EPXM = Rainfall allotted to interception storage.
 UZSN = Nominal value of the upper zone storage.
 LZSN = Nominal value of the lower zone storage.
 K3 = Index coefficient to actual evaporation.
 K24L = Fraction of groundwater recharge.
 K24EL = Fraction of groundwater lost to evapotranspiration.
 INFILTRATION = Index of soil infiltration.
 INTERFLOW = Index to alter runoff timing.
 L = Mean length of overland flow.
 SS = Average overland flow slope.
 NN = Manning's roughness coefficient for overland flow.
 IRC = Interflow recession constant.
 KV = Base flow adjustment factor.
 KK24 = Groundwater recession constant.

Table B5
Runoff Volumes for the Months of April-October
Jordan Creek Watershed

<u>Year</u>	<u>Recorded Runoff, in.</u>	<u>Simulated Runoff, in.</u>	<u>Runoff Simulated/Recorded</u>
1966	8.71	7.29	0.84
1967	17.12	20.08	1.17
1968	18.35	18.24	0.99
1969	15.89	20.63	1.30
1970	18.45	16.90	0.92
1971	29.81	29.91	1.00
1972	31.04	30.58	0.99
1973	32.21	33.72	1.05
1974	23.27	29.26	1.26
1975	29.93	34.15	1.14
1976	<u>7.24</u>	<u>8.84</u>	<u>1.22</u>
Average	23.20	24.96	1.08

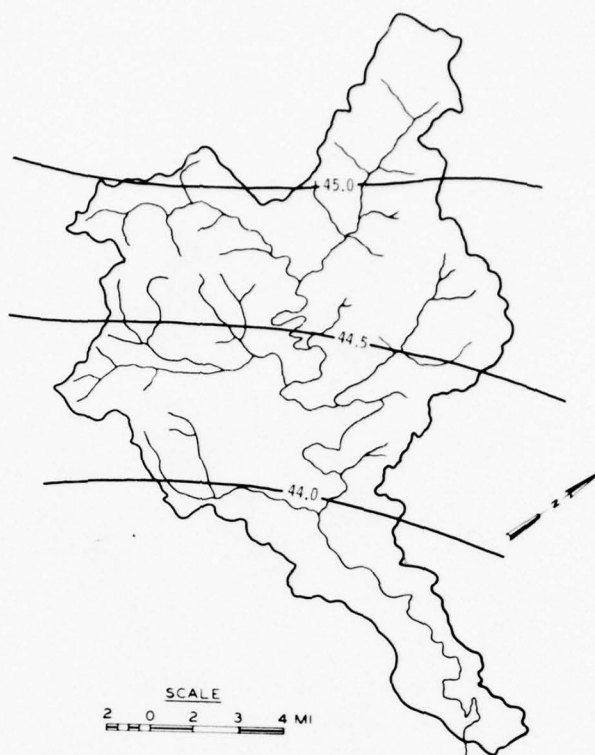


Figure B3. Isohyetal map of the Jordan Creek drainage area

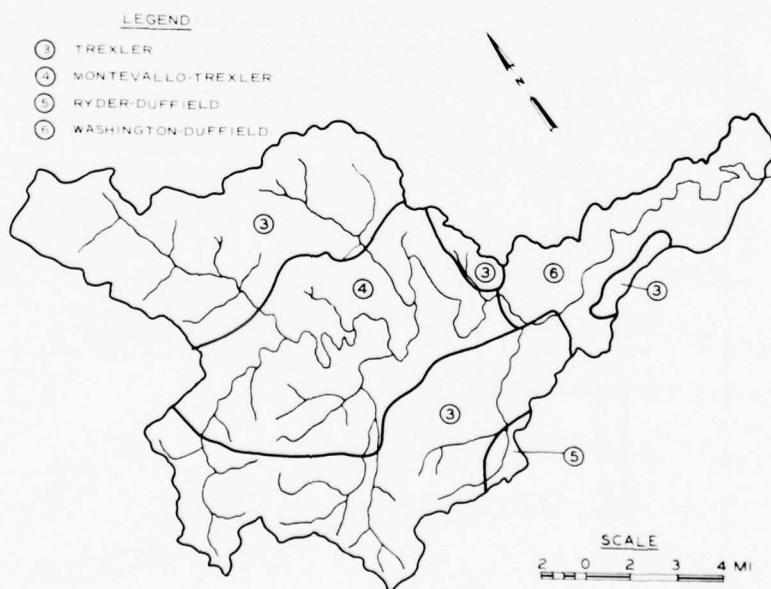


Figure B4. Schematic diagram showing soil associations for the Jordan Creek drainage basin

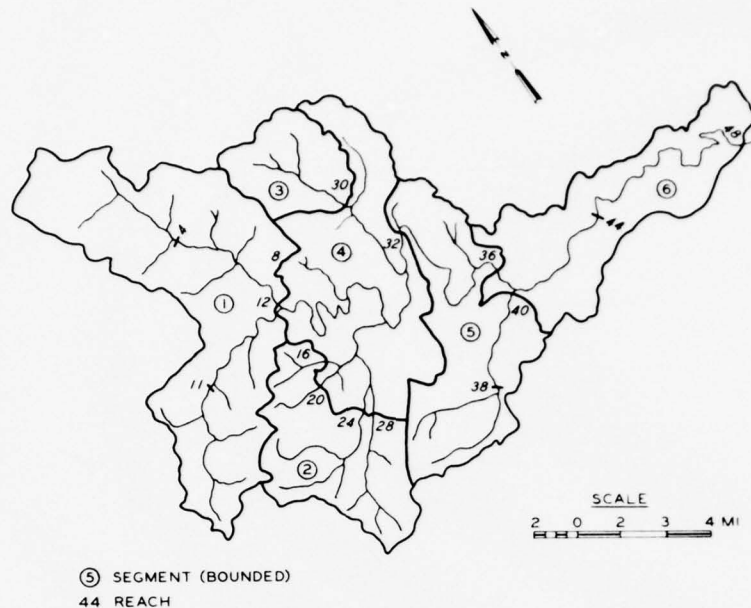


Figure B5. Schematic of the Jordan Creek drainage basin showing the boundaries of the 6 segments and the 13 reaches selected for calibrating the HSP

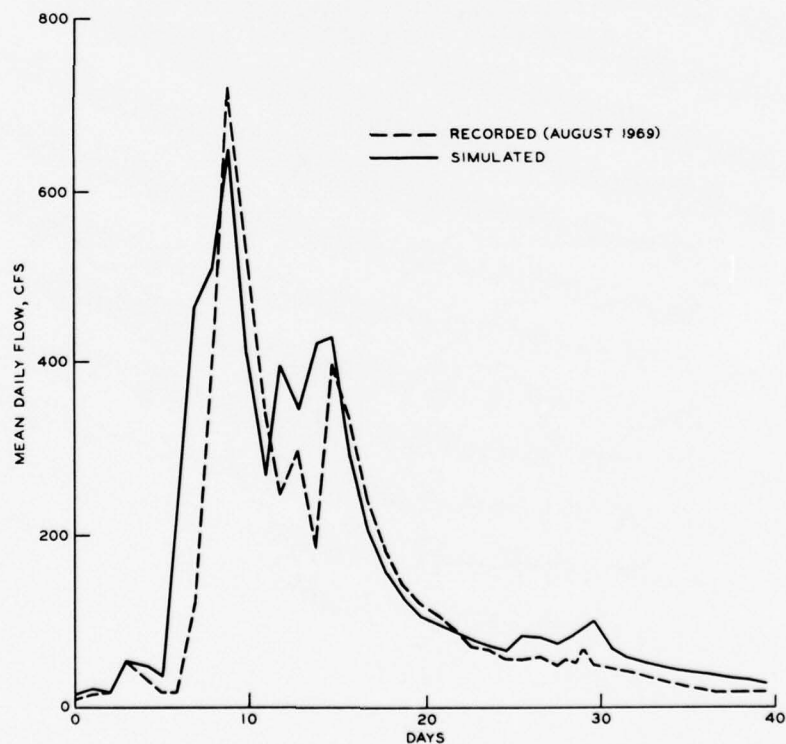


Figure B6. Calibration of the HSP for the 10-year data set for daily output at the Schnecksville streamflow gage (proposed impoundment)

CONVERSION FACTORS, U. S. CUSTOMARY TO METRIC(SI)
UNITS OF MEASUREMENT

U. S. customary units of measurement used in this appendix can be converted to metric (SI) units as follows:

<u>Multiply</u>	<u>By</u>	<u>To Obtain</u>
cubic feet per second	0.02831685	cubic metres per second
feet	0.3048	metres
inches	2.54	centimetres
miles (U. S. statute)	1.609344	kilometres
square miles	2.589988	square kilometres

APPENDIX C: INITIAL CONDITIONS, COEFFICIENTS, AND UPDATES
FOR MATHEMATICAL ECOLOGICAL SIMULATIONS

Table C1
Summary of Initial Conditions for Ecological Simulations

Parameter	1969	1973	1974
Fish			
Predators (FISH1), kg/ha	14	14	14
Planktivores (FISH2), kg/ha	2	2	2
Benthos feeders (FISH3), kg/ha	17	17	17
Water-Quality Parameters			
Algae 1 (ALGAE1), mg/l	0.008	0.008	0.008
Algae 2 (ALGAE2), mg/l	0.009	0.009	0.009
Alkalinity (ALK), mg/l	40	40	40
Benthos (BEN), mg/m ²	500-750	500-750	500-750
Ammonia (NH ₄), mg/l N	0.16	0.16	0.16
Nitrite (NO ₂), mg/l N	0.01	0.01	0.01
Nitrate (NO ₃), mg/l N	3.6	3.6	3.6
Fecal coliforms (COL), colonies/100 ml	260	260	260
Detritus (DET), mg/l	0.5	0.5	0.5
Dissolved organics (DOR), mg/l	1.0	1.0	1.0
Dissolved oxygen (DO), mg/l	12.6	12.6	12.6
Ortho-phosphate (PO ₄), mg/l P	0.01	0.01	0.01
Organic sediment (SED), mg/m ²	1000	1000	1000
Temperature (TEMP), °C	6.2	9.0	4.8
Total dissolved solids (TDS), mg/l	95	95	95
Zooplankton (ZOO), mg/l	0.02	0.02	0.02
pH (PH)	7.0	7.0	7.0

Table C2
Coefficients for Base Simulation

Parameter	Coefficient
Physical Coefficients	
Turbidity factor (TURB)*	3
Evaporative wind function (AA+BB*WIND)	
AA	0 m/(sec-mb)
BB	$1.3 \times 10^{-9} \text{ mb}^{-1}$
Mixing coefficients	
Stability parameter (GSHW)	$7.5 \times 10^{-5} - \text{sec}^{-2}$
Wind mixing coefficient (A1)	4.0×10^{-5}
Hypolimnetic diffusivity (A2)	$5.0 \times 10^{-6} \text{ m}^2/\text{sec}$
Metalimnetic coefficient (A3)	-0.5
Extinction coefficient (EXCO)	0.25 m^{-1}
Surface radiation fraction (SURFRAC)	0.6
Critical advective density (CDENS)	2.0 kg/m^3
Reaeration coefficients	
Oxygen (DMO2)	$2.04 \times 10^{-9} \text{ m}^2/\text{sec}$
Carbon dioxide (DMCO2)	$2.04 \times 10^{-10} \text{ m}^2/\text{sec}$
Stoichiometry	
O2 - NH3 (O2NH3)	3.5
O2 - NO2 (O2NO2)	1.2
O2 - Detritus (O2DET)	2.0
O2 - Respiration (O2RESP)	1.6
O2 - Algal biomass (O2FAC)	1.6
CO2 - Dissolved organics (CO2DOR)	0.2
Decay Rates	
Dissolved organics (TDORDK)	0.13 per day
Ammonia (TNH3DK)	0.15 per day
Nitrite (TNO2DK)	0.38 per day
Coliforms	
(Q10)	1.04
(TCOLDK)	1.4 per day
(Continued)	

* Acronym in parenthesis represents the variable name used in the WES version of the WQRRS model.
(Sheet 1 of 5)

Table C2 (Continued)

Parameter	Coefficient	
	ALGAE 1 I = 1	ALGAE 2 I = 2
Algae		
Chemical composition		
C	0.45	0.45
N	0.08	0.08
P	0.01	0.01
Gross production rate (TPMAX(I))	2.44 per day	2.1 per day
Temperature rate multipliers		
Lower threshold (T1)	0°C	0°C
Optimum (T2)	20°C	18°C
Optimum (T3)	27°C	22°C
Upper threshold (T4)	36°C	28°C
Half-saturation coefficients		
Carbon (PS2C02(I))	0.10 mg/l	0.10 mg/l
Nitrogen (PS2N(I))	0.018 mg/l	0.016 mg/l
Phosphorus (PS2P04(I))	0.006 mg/l	0.003 mg/l
Light (PS2L(I))	4.0 kcal/m ² /hr	8.0 kcal/m ² /hr
Respiration rate (TPRESP)	0.17 per day	0.17 per day
Settling rate (TSETL(I))	0.1 m/day	0.3 m/day
Self-shading coefficient	0.1 per m-mg/l	0.1 per m-mg/l
Zooplankton		
	<u>Coefficient</u>	
Chemical composition		
C	0.45	
N	0.08	
P	0.012	
Assimilation rate (TZMAX)	0.505 per day	

(Continued)

(Sheet 2 of 5)

Table C2 (Continued)

Parameter	Coefficient
Zooplankton (Continued)	
Temperature rate multipliers	
Lower threshold (T1)	2°C
Optimum (T2)	20°C
Optimum (T3)	26°C
Upper threshold (T4)	36°C
Assimilation efficiency (ZEFFIC)	0.65
Feeding preference	
Algae 1 (PREF(1))	0.4
Algae 2 (PREF(2))	0.6
Detritus (PREF(3))	0
Half-saturation coefficient (ZS2P)	0.2 mg/l
Mortality rate (TZMORT)	0.005 per day
Respiration rate (TZRESP)	0.2 per day
Detritus	
Chemical composition	
C	0.32
N	0.07
P	0.009
Settling rate (TDSETL)	0.15 m/day
Decay rate (TDETDK)	0.09 per day
Benthos	
Chemical composition	
C	0.47
N	0.08
P	0.011
Assimilation rate (TBMAX)	0.04 per day
Temperature rate multipliers	
Lower threshold (T1)	0°C

(Continued)

(Sheet 3 of 5)

Table C2 (Continued)

Parameter	Coefficient		
Benthos (Continued)			
Temperature rate multipliers (Continued)			
Optimum (T2)	20°C		
Optimum (T3)	26°C		
Upper threshold (T4)	36°C		
Assimilation efficiency (BEFFIC)	0.5		
Half-saturation coefficient (BS2SED)	200 mg/m ²		
Mortality rate (TBMORT)	0.005 per day		
Respiration rate (TBRESP)	0.016 per day		
	Coefficient		
	FISH 1 FISH 2 FISH 3		
	I = 1 I = 2 I = 3		
Fish			
Chemical composition			
C	0.45	0.45	0.45
N	0.08	0.08	0.08
P	0.011	0.011	0.011
Assimilation rate (TFMAX(I))	0.007 per day	0.007 per day	0.008 per day
Temperature rate multipliers			
Lower threshold (T1)	0°C	0°C	0°C
Optimum (T2)	25°C	25°C	25°C
Optimum (T3)	29°C	29°C	29°C
Upper threshold (T4)	35°C	35°C	35°C
Assimilation efficiency (FEFFIC)	0.8	0.8	0.8
Half-saturation coefficients			
Fish (FS2FSH)	10.0 kg/ha	--	--
Zooplankton--Detritus (FS2Z00)	--	0.1 mg/l	--
	(Continued)		

(Sheet 4 of 5)

(Sheet 4 of 5)

Table C2 (Concluded)

Parameter	Coefficient		
	FISH 1 I = 1	FISH 2 I = 2	FISH 3 I = 3
Fish (Continued)			
Half-saturation coefficients (Continued)			
Benthos--sediment (FS2BEN)	--	--	50 mg/l
Fraction of diet			
Sediment (F3SED)	--	--	0.45
Benthos (F3BEN)	--	--	0.55
Mortality rate (TFMAX)	0.001 per day	0.001 per day	0.001 per day
Respiration rate (TFRESP)	0.004 per day	0.004 per day	0.004 per day

Table C3
1974 Water Quality Updates

Parameter	20 Mar (79)	16 Apr (106)	8 May (128)	19 Jun (170)	12 Sep (255)	14 Oct (287)	19 Nov (323)	17 Dec (351)
ALG 1, mg/l				Assume = 0				
ALG 2, mg/l				Assume = 0				
ALK, mg/l				ALK = 38.9(FLOW) ^{-0.269}				
BOD, mg/l				Assume = 1				
NH ₄ -N, mg/l	0.05	0.17	0.15	0.22	0.03	0.04	0.04	0.06
NO ₂ -N, mg/l				Assume = 0.013				
NO ₃ -N, mg/l	1.8	4.2	2.3	2.7	1.3	3.4	2.7	4.3
COL, colonies/100 ml	370	260	3	1100	160	230	130	370
DET-C, * mg/l	--	0.5	1.4	--	--	1.9	7.3	2.8
DO, mg/l				Assume 100 percent saturation				
PO ₄ -P, mg/l	0.01	0.01	0.0**	0.03	0.03	0.01	0.01	0.04
TEMP, °C			Generated from air temperature residual					
TDS, † mg/l	105	105	105	117	120	127	132	120
ZOO, mg/l				Assume = 0				
PH	7.2	6.4	8.7	6.3	7.5	6.8	8.2	6.3

Note: Julian day is given in parentheses after date.

* DET = TOC.

** Replaced by Mean Annual Value.

† TDS = 0.6 (Sp Cond).

Table C4
1973 Water Quality Updates

Parameter	2 Apr (92)	28 Apr (118)	23 May (143)	12 Jun (163)	18 Jul (199)	14 Aug (226)	12 Sep (255)	29 Oct (302)	14 Nov (318)	5 Dec (339)
ALG 1, mg/l					Assume = 0					
ALG 2, mg/l					Assume = 0					
ALK, mg/l					ALK = 38.9(FLOW) ^{-0.269}					
BOD, mg/l					Assume = 1					
NH ₄ -N, mg/l	0.13	0.02	0.22*	0.04*	0.05*	0.10*	0.03*	0.06	0.08	0.11
NO ₂ -N, mg/l					Assume = 0.0013					
NO ₃ -N, mg/l	2.7	2.5	2.7	3.0*	2.8*	1.7*	1.3*	2.3	3.8	2.7
COL, colonies/100 ml	2400	600	440	1400	200	110	160	340	220	390
DET-C,** mg/l	10.0	7.0	4.0	--	0+	0+	0+	--	6.0	2.0
DO, mg/l					Assume 100 percent saturation					
PO ₄ -P, mg/l	0.09	0.01	0.02	0.01*	0.01*	0.03*	0.03*	0.02	0+	0.02
TEMP, °C					Generated from air temperature residual					
TDS,†† mg/l	84	96	102	108	114	126	120	144	126	105
ZOO, mg/l					Assume = 0					
PH	6.9	7.5	7.1	9.1	8.8	8.8	7.5	7.0	7.3	6.8

Note: Julian day is given in parentheses after date.

* Dissolved form only.

** DET = TOC.

† Replaced by Mean Annual Value.

†† TDS = 0.6 (Sp Cond).

Table C5
1969 Mean Annual Water Quality Updates

Parameter	Update
ALG 1, mg/l	0
ALG 2, mg/l	0
ALK, mg/l	37
BOD, mg/l	1.0
NH ₄ -N, mg/l	0.1
NO ₂ -N, mg/l	0.013
NO ₃ -N, mg/l	3.4
COL, colonies/100 ml	280
DET-C,* mg/l	3.9
DO, mg/l	Assume 100 percent saturation
PO ₄ -P, mg/l	0.018
TEMP, °C	Generated from air temperature residual
TDS,** mg/l	115
ZOO, mg/l	0
PH	7.4

* DET = TOC.

** TDS = 0.6 (Sp Cond).

APPENDIX D: COEFFICIENT REFERENCES

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